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Sediment Classification Methods Compendium

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Agency

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SEDIMENT CLASSIFICATION METHODS COMPENDIUM

by

U.S. Environmental Protection Agency

Portions of this document were prepared by
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- Gerald Ankley, Anthony R. Carlson, Phillip M. Cook, Wayne S. Davis, Catherine Krueger, Janet Lamberson, Henry Lee II, Richard C. Swartz, Nelson Thomas, and Christopher S. Zarba (U.S. EPA)
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CHAPTER 1. INTRODUCTION

1.0 BACKGROUND

Sediment management issues are of importance to many programs within the U.S. Environmental Protection Agency (EPA). The ability to assess sediment quality in a technically reliable and legally defensible manner is necessary for effective sediment management. In the summer of 1988, the U.S. EPA Office of Water Regulations and Standards (OWRS) formed two committees to identify, coordinate, and provide guidance on activities relating to the assessment and management of sediments contaminated with toxic chemicals: a Sediment Oversight Technical Committee and a Sediment Oversight Steering Committee. The goal of these committees is to facilitate decisions made at various stages in the management process such as assessing sediment contamination, deciding on the need for and type of management action, and evaluating types of remediation. This document, prepared by the Sediment Oversight Technical Committee, describes the various methods used to assess sediment quality.

A number of approaches can be used to assess sediment contamination. Many past approaches were based on comparing chemical concentrations in contaminated areas to those in reference areas, and did not directly consider biological effects. More recent approaches to evaluating sediment quality have focused on determining relationships between sediment contaminant concentrations and adverse biological effects. These approaches may be applied to a variety of regulatory decisions, including identification of problem areas, establishment of cleanup goals, development of discharge and dumping permit criteria, and determination of monitoring requirements.

2.0 OBJECTIVE

This compendium is essentially an "encyclopedia" of methods that are used to assess chemically contaminated sediments. It contains a description of each method, associated advantages and limitations, and existing applications. It is intended to serve as a common frame of reference to answer the "how clean is clean" question for particular sediments (i.e., does sediment contamination exist to a degree that warrants the evaluation of need for further action?). Some of the methods in this compendium can also be used as part of subsequent regulatory or remedial actions. It should be pointed out that these methods are not at an equal stage of development, and certain ones (or combinations) are more appropriate for specific management actions than are others. This document is not meant to provide guidance on which method(s) to apply for specific situations, nor how they can be used together as part of a decision-making framework. Such guidance will be forthcoming and will likely include both chemical and biological methods in a tiered type of framework.

3.0 OVERVIEW

The sediment quality assessment methods described in this report can be classified into two basic types: numeric or descriptive (Table 1-1). Numeric methods are chemical-specific and can be used to generate numerical sediment quality criteria. Descriptive methods are not chemical-specific and cannot be used alone to generate numerical sediment quality criteria for particular chemicals.

In addition, some of the approaches described in this report comprise at least two methods and can be classified as combination approaches (Table 1-1). For example, the Sediment Quality Triad (Triad) and Apparent Effects Threshold (AET) approaches employ bulk sediment toxicity testing, benthic community structure analysis, and concentrations of sediment contaminants. The Triad is both descriptive and numeric, depending on its use. Typically,

TABLE 1-1. SEDIMENT QUALITY ASSESSMENT METHODS

Method (Chapter)	Type Numeric	 Descriptive	 Combination	Concept
Bulk Sediment Toxicity (2.0)		■		Test organisms are exposed to sediments which may contain unknown quantities of potentially toxic chemicals. At the end of a specified time period, the response of the test organisms is examined in relation to a specified biological endpoint.
Spiked-Sediment Toxicity (3.0)	■			Dose-response relationships are established by exposing test organisms to sediments that have been spiked with known amounts of chemicals or mixtures of chemicals.
Interstitial Water Toxicity (4.0)	■			Toxicity of interstitial water is quantified and identification evaluation procedures are applied to identify and quantify chemical components responsible for sediment toxicity. The procedures are implemented in three phases to characterize interstitial water toxicity, identify the suspected toxicant, and confirm toxicant identification.
Equilibrium Partitioning (5.0)	■			A sediment quality value for a given contaminant is determined by calculating the sediment concentration of the contaminant that would correspond to an interstitial water concentration equivalent to the U.S. EPA water quality criterion for the contaminant.
Tissue Residue (6.0)	■			Safe sediment concentrations of specific chemicals are established by determining the sediment chemical concentration that will result in acceptable tissue residues. Methods to derive unacceptable tissue residues are based on chronic water quality criteria and bioconcentration factors, chronic dose-response experiments or field correlations, and human health risk levels from the consumption of freshwater fish or seafood.
Freshwater Benthic Community Structure (7.0)		■		Environmental degradation is measured by evaluating alterations in freshwater benthic community structure.
Marine Benthic Community Structure (8.0)		■		Environmental degradation is measured by evaluating alterations in marine benthic community structure.
Sediment Quality Triad (9.0)	■	■	■	Sediment chemical contamination, sediment toxicity, and benthic infauna community structure are measured on the same sediment. Correspondence between sediment chemistry, toxicity, and biological effects is used to determine sediment concentrations that discriminate conditions of minimal, uncertain, and major biological effects.
Apparent Effects Threshold (10.0)	■		■	An AET is the sediment concentration of a contaminant above which statistically significant biological effects (e.g., amphipod mortality in bioassays, depressions in the abundance of benthic infauna) would always be expected. AET values are empirically derived from paired field data for sediment chemistry and a range of biological effects indicators.

TABLE 1-1. (Continued)

International Joint Commission (II.0)[a]	Contaminated sediments are assessed in two stages: 1) an initial assessment that is based on macrozoobenthic community structure and concentrations of contaminants in sediments and biological tissues, and 2) a detailed assessment that is based on a phased sampling of the physical, chemical, and biological aspects of the sediment, including laboratory toxicity bioassays.

[a] The IJC approach is an example of a sequential approach, or "strategy" combining a number of methods for the purpose of assessing contaminated sediments in the Great Lakes.

the Triad approach has been used in a descriptive manner to identify contaminated sediments. However, it has also been used to generate criteria for several chemical contaminants. The International Joint Commission (IJC) approach presented at the end of this document (Chapter 11) is an example of a sequential approach, or "strategy," combining a number of methods for the purpose of assessing contaminated sediments in the Great Lakes.

Each sediment quality assessment method is presented as a separate chapter. Each chapter is structured identically, as indicated in Table 1-2, to facilitate comparisons among the various methods. Authors are listed at the beginning of each chapter. A general description, application, usefulness, and status of the method is then presented. A list of references cited is provided at the end of each chapter.

TABLE 1-2. STRUCTURE OF SEDIMENT QUALITY ASSESSMENT METHOD CHAPTERS

- 1.0 SPECIFIC APPLICATIONS
  - 1.1 CURRENT USE
  - 1.2 POTENTIAL USE
- 2.0 DESCRIPTION
  - 2.1 DESCRIPTION OF METHOD
    - 2.1.1 Objectives and Assumptions
    - 2.1.2 Level of Effort
      - 2.1.2.1 Type of Sampling Required
      - 2.1.2.2 Methods
      - 2.1.2.3 Types of Data Required
      - 2.1.2.4 Necessary Hardware and Skills
    - 2.1.3 Adequacy of Documentation
  - 2.2 APPLICABILITY OF METHOD TO HUMAN HEALTH, AQUATIC LIFE, OR WILDLIFE PROTECTION
  - 2.3 ABILITY OF METHOD TO GENERATE NUMERICAL CRITERIA FOR SPECIFIC CHEMICALS
- 3.0 USEFULNESS
  - 3.1 ENVIRONMENTAL APPLICABILITY
    - 3.1.1 Suitability for Different Sediment Types
    - 3.1.2 Suitability for Different Chemicals or Classes of Chemicals
    - 3.1.3 Suitability for Predicting Effects on Different Organisms
    - 3.1.4 Suitability for In-Place Pollutant Control
    - 3.1.5 Suitability for Source Control
    - 3.1.6 Suitability for Disposal Applications
  - 3.2 GENERAL ADVANTAGES AND LIMITATIONS
    - 3.2.1 Ease of Use
    - 3.2.2 Relative Cost
    - 3.2.3 Tendency to be Conservative
    - 3.2.4 Level of Acceptance
    - 3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities
    - 3.2.6 Level of Effort Required to Generate Results
    - 3.2.7 Degree to Which Results Lend Themselves to Interpretation
    - 3.2.8 Degree of Environmental Applicability
    - 3.2.9 Degree of Accuracy and Precision

TABLE 1-2. (Continued)

4.0 STATUS

- 4.1 EXTENT OF USE
- 4.2 EXTENT TO WHICH APPROACH HAS BEEN FIELD-VALIDATED
- 4.3 REASONS FOR LIMITED USE
- 4.4 OUTLOOK FOR FUTURE USE AND AMOUNT OF DEVELOPMENT YET NEEDED

5.0 REFERENCES

CHAPTER 2. BULK SEDIMENT TOXICITY TEST APPROACH

Nelson Thomas
U.S. Environmental Protection Agency
Environmental Research Laboratory
6201 Congdon Blvd.
Duluth, MN 55804
(218) 720-5702

Janet Lamberson and Richard C. Swartz
U.S. Environmental Protection Agency
Environmental Research Laboratory - N. Pacific Division
Hatfield Marine Science Center
Newport, OR 97365
(503) 867-4031

In the bulk sediment toxicity approach, test organisms are exposed in the laboratory to sediments that were collected in the field. A specific biological endpoint is used to assess the response of the organisms to the sediments (i.e., to measure sediment toxicity). The bulk sediment toxicity approach is a descriptive method and cannot be used by itself to generate sediment quality criteria.

1.0 SPECIFIC APPLICATIONS

1.1 Current Use

Sediment toxicity testing has been applied in the following ways in dredged material disposal permit and other regulatory programs (U.S. EPA and U.S. Army COE 1977).

- To determine potential biological hazards of dredged material intended for disposal in an aquatic environment

- To evaluate the effectiveness of various dredged material management actions

- To indicate spatial distribution of toxicity in contaminated areas, relative degree of toxicity, and changes in toxicity along a gradient of pollution or with respect to distance from pollutant sources (Swartz et al. 1982, 1985b)

- To reveal temporal changes in toxicity (i.e., by sampling the same locations over time or by assaying layers of buried sediment in core samples) (Swartz et al. 1986)

- To reveal "hot spots" of contaminated sediment for further investigation (Chapman 1986a)

- To rank sediments based on toxicity to benthic organisms and to define boundaries of small or large problem areas for cleanup of contaminated sediment.

Bulk sediment toxicity testing integrates interactions among complex mixtures of contaminants that may be present in the field. Many classes of chemical contaminants, including metals, PAHs, PCBs, dioxins, and chlorinated pesticides, can contribute to toxicity in effluents and sediments (Chapman et al. 1982). The bulk sediment toxicity test measures the total toxic effect of all contaminants, regardless of physical and chemical composition.

1.2 Potential Use

By itself, bulk sediment toxicity testing cannot generate chemical-specific toxic effects data but does determine toxicity. However, used in conjunction with toxicity identification evaluation procedures such as those described in Chapter 4, 9, and 10, bulk sediment toxicity testing could help identify causal toxicants. The procedure must be combined with other methods of estimating sediment quality in order to generate sediment quality criteria, such as the Sediment Quality Triad (Triad) (Chapman 1986b; Chapman et al. 1987; see Chapter 9), and the Apparent Effects Threshold (AET) approach (Tetra Tech 1986; PTI 1988; see Chapter 10). Bulk sediment toxicity will be most valuable in verification of other methods used to develop sediment quality criteria. This method is also useful in determining acceptability for disposal options.

2.0 DESCRIPTION

2.1 Description of Method

The toxicological approach involves exposing test organisms to sediments. The chemical composition of the sediments, which may be complex, need not be known. At the end of a specified time period, the response of the test organisms is examined in relation to a specified biological endpoint (e.g., mortality, growth, reproduction, cytotoxicity, alterations in development or respiration rate). Results are then compared with control and reference sediment results to estimate sediment toxicity.

2.1.1 Objectives and Assumptions--

The objective of this approach is to derive toxicity data that can be used to predict whether the test sediment will be harmful to benthic biota. It is assumed that the behavior of chemicals in test sediments in the laboratory is similar to that in natural *in situ* sediments. The effects of various interactions (e.g., synergism, additivity, antagonism) among chemicals in the field or in dredged materials can be predicted from laboratory results without measuring total or bioavailable concentrations of potentially hundreds of contaminants in the test sediment (Swartz et al. in press) and without *a priori* knowledge of specific pathways of interaction between sediments and test organisms (Kemp and Swartz in press). In that one of the strengths of this test is to integrate effects of all contaminants, the effect of individual contaminants cannot be determined, therefore limiting its use in source control. The method can be used for all classes of sediments and any chemical contaminant, but not to answer cause-and-effect questions.

2.1.2 Level of Effort--

Implementation of this procedure requires a moderate amount of laboratory effort. A variety of toxicity test procedures (see Section 2.1.2.2) have been developed and are generally fairly straightforward and well documented.

2.1.2.1 Type of Sampling Required--It is recommended that bulk sediments be collected for analysis of total solids, grain size, acid volatile sulfide, and total and dissolved organic carbon. Bulk and interstitial concentrations of chemicals of interest in the test sediment can be determined in subsamples of the sediment added to the toxicity test chambers to enhance the interpretation of toxicity results. However, methods for sampling interstitial water have not been standardized. Sediment variables such as pH and Eh should also be monitored.

2.1.2.2 Methods--There currently are several bulk sediment toxicity tests under ballot by the American Society of Testing Materials (ASTM). The most commonly used of these partial life cycle tests feature freshwater chironomid species (*Chironomus tentans, Chironomus riparius*), the freshwater/estuarine amphipod *Hyalella azteca*, and the marine amphipod *Rhepoxynius abronius*. Brief generalized descriptions of these tests are given below.

Bulk sediment toxicity tests with the two freshwater chironomid species are functionally very similar, differing only in the age of the organisms with which the test is initiated, and the duration of the test. Both *C. tentans* and *C. riparius* are available from various aquatic toxicology laboratories and commercial sources, and both species are easily cultured in a laboratory setting. Toxicity tests are initiated by adding *C. riparius* <3 days old or *C. tentans* 10-14 days old (second instar) to test chambers that contain bulk sediment with overlying water in various ratios (e.g., 6 water:1 sediment; Giesy et al. 1988). The length of the test also varies with the biological endpoint of interest and the species used. If the biological endpoint of interest is growth and survival of the larvae, the test is terminated after 10-14 days by sieving the *C. riparius* or *C. tentans* from the sediment. It also is possible to conduct the test until the adults emerge, which will occur (depending upon temperature) in around 30 days for *C. riparius* and at 20-25 d for *C. tentans*. More detailed descriptions of toxicity test procedures with *C. riparius* and *C. tentans* are given by Adams et al. (1985), Nebeker et al. (1984), Giesy et al. (1988), and Ingersoll and Nelson (1989).

Partial life cycle toxicity tests with the freshwater/estuarine amphipod *H. azteca* and bulk sediments have been conducted in a number of laboratories. *H. azteca* are available from various aquatic toxicology laboratories and commercial sources, and can be easily cultured in a laboratory setting. Toxicity tests are initiated by adding juveniles <7 days old to test chambers that contain bulk sediment with overlying water in various ratios (e.g., 4 water:1 sediment; Ingersoll and Nelson 1989). The length of the test can range from ≤10 days (short-term partial life cycle) to 30 days (long-term partial life cycle) (Nebeker et al. 1984; Ingersoll and Nelson 1989). Depending upon the length of the test, biological endpoints include survival, behavior, growth, and reproduction. More detailed descriptions of toxicity test procedures are given by Nebeker et al. (1984), Nebeker and Miller (1988), and Ingersoll and Nelson (1989).

Partial life cycle toxicity tests with the marine amphipod *Rhepoxynius abronius* and bulk sediments have been used routinely for some time. *R. abronius* and bulk sediments generally are collected from the field and acclimated to laboratory conditions for some time (<14 days) before toxicity testing. The tests are initiated by adding large immature and adult amphipods to test chambers that contain bulk sediment with overlying water in various ratios. The length of the test generally is $\geq$10 days and the biological responses monitored consist of behavioral effects (e.g., emergence from sediment) and mortality. More detailed descriptions of the toxicity test procedures are given by Swartz et al. (1979, 1985), DeWitt et al. (1988), and Robinson et al. (in press). Other ASTM candidate species for marine toxicity tests are *Eohaustorius estuarius*, *Ampelisca abdita*, and *Grandidierella japonica*.

2.1.2.3 Types of Data Required--The physical and chemical data described in Section 2.1.2.1 are needed to interpret the test results. Biological data required, which vary by test, may include mortality and various sublethal effects (e.g., changes in growth, reproduction, respiration rate, behavior, or development). These data can be compared to control and reference data to determine the occurrence of biological effects. Dilution experiments in which uncontaminated sediment is added to test sediment collected from the field can be used to calculate LC50 values, EC50 values, no-effect concentrations, and lowest observable-effect concentrations. However, standardized techniques with dilution (i.e., by sediment of similar physical-chemical properties) have not been developed.

2.1.2.4 Necessary Hardware and Skills-- In general, only readily available and inexpensive field and laboratory equipment is needed, procedures are fairly simple and straightforward, and a minimum of training is needed to detect endpoints through toxicity tests. Interpretation of the toxicity (chemical and biological) data requires a higher degree of skill and training. Chemical sampling methods are generally simple and routine, although analyses of chemical samples requires specialized training and equipment. Some biological effects tests also require specialized training, handling, and facilities.

2.1.3 Adequacy of Documentation--

Various sediment toxicity test procedures have been developed and are well documented for testing field sediments (Lamberson and Swartz 1988; Swartz 1987). However, methods must be better standardized through use and intercalibrated among laboratories, and most methods need better field validation.

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

The bulk sediment toxicity test approach is suitable only for protection of aquatic life. Sediment toxicity test procedures incorporate a direct measure of sediment biological effects, and can be used to predict biological effects of contaminated sediments prior to approval of state or federal permits. They can be used to assess the toxicity of sediments in the natural environment and to predict the effects of these sediments on resident aquatic life. Combined with other approaches, such as in the AET and the Triad approaches, they can be used to establish sediment quality criteria.

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

The bulk sediment toxicity test approach cannot be used by itself to generate sediment quality criteria, but must be combined with chemical measurements and other data to generate information on effects of individual contaminants. Both the Triad and the AET approaches rely on bulk sediment toxicity data to derive numerical criteria. Bulk sediment toxicity tests in conjunction with sediment quality criteria derived from Equilibrium Partitioning (U.S. EPA 1980) (see also Chapter 5 herein) can also be used in assessments of potentially contaminated sediments.

3.0 USEFULNESS

3.1 Environmental Applicability

3.1.1 Suitability for Different Sediment Types--

The sediment toxicity test approach is suitable for any type of sediment. In some cases, the physical or chemical properties of the test sediment, such as salinity or grain size, may limit the selection of organisms that can be used for testing, and may also affect interpretation of the data (Ott 1986; DeWitt in preparation). Appropriate controls for sediment properties may be necessary to discriminate chemical toxicity from conventional effects. In establishing sediment quality criteria, the effects of toxic features of the sediment itself, such as grain size, must be recognized (DeWitt et al. 1988). Data can be normalized to such factors as organic carbon, and thus can be applied to any sediment. However, normalization techniques are in the developmental stage (see Equilibrium Partitioning, Chapter 5).

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

This is the only approach currently available that directly measures biological effects of all classes of chemicals, including the combined interactive (additive, synergistic, antagonistic) toxic effects among individual chemicals in mixtures of contaminants usually found in field sediments (Plesha et al. 1988; Swartz et al. in press). Bioaccumulative chemicals can be evaluated if the length of the test is extended to ensure adequate exposure of the test organism.

3.1.3 Suitability for Predicting Effects on Different Organisms--

Theoretically, any organism can be used in sediment toxicity testing. To protect a biological community and to predict the effects of contaminated sediments on different organisms, test organisms should be selected on the basis of their sensitivity to contaminants, their ability to withstand laboratory handling, and their ability to survive in control and reference treatments (Swartz et al. 1985a). In tests to determine the effects of contaminated sediments on a particular biological community, the test species selected should be among the most sensitive found in the community of interest, or should be comparably sensitive. Test species should include more than one type of organism to ensure a range of sensitivity to various types of contaminants.

3.1.4 Suitability for In-Place Pollutant Control--

Sediment toxicity testing can be used directly to monitor in-place pollution. As noted above in Section 1.1, sediment toxicity testing can be used to determine the extent of the problem area, to monitor temporal and spatial trends, to detect the presence of unsuspected "hot spots," to assess the need for remedial actions, and to monitor changes in toxicity after remedial action is taken. Such tests can also be used as a cost-effective and rapid screening tool for in-place pollutant reconnaissance surveys, and in *a priori* simulations of proposed remedial actions to test effectiveness of capping or other remedial alternatives.

3.1.5 Suitability for Source Control--

Bulk field sediment toxicity testing can be used to identify suspected sources of sediment pollution. Field reconnaissance surveys can reveal "hot spots" in the vicinity of sources, and a map showing contours of sediment toxicity values can reveal gradients that identify point and nonpoint sources (Swartz et al. 1982). Toxicity testing cannot be used by itself to verify reductions in mass loading of chemicals that might be expected as a result of source control. The biological effects of source control can be represented, however, through the use of bulk sediment toxicity testing.

3.1.6 Suitability for Disposal Applications--

Bulk sediment toxicity testing can be used in regulatory programs to determine the toxicity of material prior to disposal. The potential hazard to benthic organisms at the disposal site (which is determined by making comparisons with the "reference" sediments collected near the disposal site) can be predicted from laboratory toxicity test results. Sediment toxicity tests can also be used to monitor conditions at the disposal site both before and after a disposal operation.

3.2 General Advantages and Limitations

3.2.1 Ease of Use--

Most sediment toxicity test procedures are simple to use, requiring limited expertise and standard inexpensive laboratory equipment. Only a few sublethal effects tests require specialized training. Field sampling requires only readily available equipment and standard procedures.

3.2.2 Relative Cost--

The cost of individual laboratory toxicity tests as well as field sampling is low because of the limited expertise and inexpensive equipment requirements. Costs generally range from $150 to $500 per replicate. Laboratory sediment toxicity testing is a comparatively inexpensive and cost-effective method of monitoring the field distribution of sediment toxicity, because it integrates the effects of all toxic contaminants, does not require individual chemical measurements, and does not require time-consuming analysis of benthic community structure.

3.2.3 Tendency to be Conservative--

Sediment toxicity tests can be made as sensitive or as conservative (i.e., environmentally protective) as necessary through selection of biological endpoints and species of test organism. Reliance on mortality as an endpoint may be underprotective, while some sublethal endpoints (e.g., enzyme inhibition) may be overprotective.

3.2.4 Level of Acceptance--

Bulk sediment toxicity testing is widely accepted by the scientific and regulatory communities, and has been tested and contested in court. Field sediment toxicity test results have been widely published in peer-reviewed journals, and have been incorporated into other measures of sediment quality such as the AET and the Triad approaches. Standard guides for sediment toxicity testing are being developed by ASTM. Field sediment toxicity testing is incorporated into most dredged material disposal regulatory programs. Toxicity testing in general has long been the basis for water quality criteria, dredged material testing, effluent testing, and discharge monitoring.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

Sediment toxicity test methods are easily implemented by laboratories with typical equipment using inexpensive glassware and procedures requiring little specialized training, although the interpretation of some sublethal biological endpoints may require some degree of training and experience. Field sediment sample collection procedures are routine.

3.2.6 Level of Effort Required to Generate Results--

This procedure consists of field sampling and a laboratory toxicity tests. Compared to an extensive survey of chemical concentrations or benthic community structure analysis, the level of effort is relatively small.

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

Biological responses to toxic sediment can be easily interpreted. Generally data fit "pass-fail" criteria (i.e., the result is either above or below a predetermined acceptance level) or the result is statistically compared to control and reference results to determine whether there is a toxic effect. Little "expert" guidance is required for interpretation of the results.

3.2.8 Degree of Environmental Applicability--

As noted in Section 3.1, the sediment toxicity test approach is applicable to a wide range of environmental conditions and sediment types. The effects of various sediment properties such as grain size and organic content can be experimentally addressed with appropriate uncontaminated controls.

3.2.9 Degree of Accuracy and Precision--

Since the sediment toxicity test is a laboratory-controlled experiment, results have a high degree of accuracy, precision, and repeatability. The procedure produces a direct biological response data set for individual sediment samples.

4.0 STATUS

4.1 Extent of Use

Sediment toxicity tests are widely used in research and regulatory programs in both marine and freshwater systems, as described in Section 1.1. Sediment toxicity tests are also incorporated into evaluation of applications for dredged material disposal permits, and are used to assess the toxicity of sediments subject to regulatory decisions. Bulk sediment toxicity tests are also used to investigate the mechanisms of sediment toxicity to benthic organisms (Kemp and Swartz 1989).

4.2 Extent to Which Approach Has Been Field-Validated

Field validation of bulk sediment toxicity testing includes several publications in the peer-reviewed literature (Chapman 1986b; Plesha et al. 1988; Swartz et al. 1982, in press). As more data become available, results can be compared with available information on contaminant concentrations in sediment in areas where biological effects have been observed. The effects of interactions among contaminants, as well as the effects of nonchemical sediment variables must be taken into consideration when attempts are made at field validation. As noted in Section 2.1.3, better field validation of predicted effects is needed.

4.3 Reasons for Limited Use

Bulk sediment toxicity testing has been widely used in research and regulatory programs (see above).

4.4 Outlook for Future Use and Amount of Development Yet Needed

The outlook for future use of sediment toxicity tests is promising where direct measurement o biological effects of toxicants in sediments is desired, especially where the effects of chemical interactions is of interest. Development of biological testing methods should continue, and more emphasis should be placed on the development of procedures to measure chronic effects. Methods should be compared and standardized among laboratories, and results should be field-validated to establish their ability to predict biological effects in the field. As more toxicity tests are conducted and the results subject to a quality assurance review, results should be compiled in a central database so that comparisons can be made among species, methods, and laboratories.

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CHAPTER 3. SPIKED-SEDIMENT TOXICITY TEST APPROACH

Janet Lamberson and Richard C. Swartz
U. S. Environmental Protection Agency
Environmental Research Laboratory - N. Pacific Division
Hatfield Marine Science Center
Newport, OR 97365
(503) 867-4031

The toxicological approach to generating sediment quality criteria uses concentration-response data from sediments spiked in the laboratory with known concentrations of contaminants to establish cause-and-effect relationships between chemicals and adverse biological responses (e.g, mortality, reductions in growth or reproduction, physiological changes). Individual chemicals or other potentially toxic substances can be tested alone or in combination to determine toxic concentrations of contaminants in sediment. This approach can be used to generate sediment quality criteria or to validate sediment quality criteria generated by other approaches.

1.0 SPECIFIC APPLICATIONS

1.1 Current Use

The spiked-sediment toxicity test approach is still in the research stage. Although the procedures resemble those used to generate water quality criteria, the influence of variable properties of sediment makes the generation of sediment quality criteria values much more complex.

Where LC50 values and chronic effects data are available for chemicals in sediments (see Section 2.3), they can be used to identify concentrations of chemicals in sediment that are protective of aquatic life. The predictive value of sediment quality criteria generated by this approach should be tested by comparing them with field data on chemical concentrations in natural sediments and observed biological effects. Interim laboratory-derived criteria, however, can be implemented prior to field validation.

1.2 Potential Use

This method can be used to empirically address the problem of interactions among complex mixtures of contaminants that are almost always present in the field. Chemical-specific data can be generated for a wide variety of classes of chemical contaminants, including metals, PAHs, PCBs, dioxins, and chlorinated pesticides. Both acute and chronic criteria can be established, and the approach is applicable to both marine and freshwater systems (Tetra Tech 1986; Battelle 1988). However, unless the sediment factor that normalizes for bioavailability is known, this procedure must be applied to every sediment (i.e., a value derived for one sediment may not be applied with predictable results to another sediment with different properties).

2.0 DESCRIPTION

2.1 Description of Method

The toxicological approach involves exposing test organisms to sediments that have been spiked with known quantities of potentially toxic chemicals or mixtures of compounds. At the end of a specified time period, the response of the test organism is examined in relation to a biological endpoint (e.g., mortality, growth, reproduction, cytotoxicity, alterations in development or respiration rate). Results are then statistically compared with results from control or reference sediments to identify toxic concentrations of the test chemical.

2.1.1 Objectives and Assumptions--

The objective of this approach is to derive concentration-response values in the laboratory that can be used to predict concentrations of specific chemicals that would be harmful to resident biota under field conditions. The effects of interactions (i.e., synergism, additivity, antagonism) among chemicals in the field can be predicted from laboratory results with sediments spiked with combinations of chemicals. The method can be used for all classes of sediments and any chemical contaminant. The bioavailable component of contaminants in sediment can be determined by this method, and *a priori* knowledge of specific pathways of interaction between sediments and test organisms is not necessary. Any method of expressing the bioavailability of contaminants in sediment can be used in conjunction with sediment toxicity tests, including the "free" interstitial concentration and normalizations to organic carbon and other sediment properties.

Data generated by this method may be difficult to interpret if the normalizing factor for bioavailability is unknown. If the normalization factor is known, this method can be used to validate sediment quality criteria generated by other approaches. It is assumed that laboratory results for a given sediment and overlying water represent biological effects of similar sediments in the field, and that the behavior of chemicals in spiked sediments is similar to that in natural *in situ* sediments.

2.1.2 Level of Effort--

Implementation of this procedure requires a moderate to considerable amount of laboratory effort. The various toxicity test procedures that have been developed are generally straightforward and well documented (reviewed by Lamberson and Swartz 1988; Nebeker et al. 1984; Swartz 1987; Tetra Tech and EVS 1986). However, many individual tests would be required to generate an extensive database of sediment quality values for a large number of chemicals, chemical combinations, and sediment types.

2.1.2.1 Type of Sampling Required--Collection of sediments from the field is required. Depending on the particular study objectives, the sediments may be clean (i.e., uncontaminated) sediments from a control area, uncontaminated reference sediments for comparison with similar contaminated sediments, or contaminated sediments to be spiked with known concentrations of chemicals in a test for interactions among contaminants. Sufficient sediment must be collected to provide samples for chemical analysis, spiking, and reference or controls (i.e., sediment for statistical comparison with spiked sediment). Depending upon the experimental design, the following controls may be required: sediment from the collection site for test animals, positive controls with a reference toxicant, carrier controls, and controls for natural sediment features that may affect test animals (e.g., grain size distribution).

2.1.2.2 Methods--Various methods of adding chemicals to sediment (i.e., spiking sediments) have been used. In general, the chemical is either added to the sediment and mixed in (Francis et al. 1984; Swartz et al. 1986b 1988; Birge et al. 1987), or added to the overlying water (Hansen and Tagatz 1980; Kemp and Swartz 1988) or to a sediment slurry (Landrum et al. in press; Oliver 1984; Schuytema et al. 1984) and allowed to equilibrate with the sediment. Sediments are spiked with a range of concentrations to generate LC50 data or to determine a minimum concentration at which biological effects are observed.

The effect of sediment contaminants on benthic biota is determined either by exposing known numbers of individual benthic test organisms to the sediment for a specific length of time (e.g., Swartz et al. 1985), or by exposing larvae of benthic species to the sediment in flowing natural waters (Hansen and Tagatz 1980). Biological responses are determined at the end of the test period using response criteria that include mortality, changes in growth or reproduction, behavioral or physiological alterations, or differences in numbers and species of larvae that become established in contaminated vs. control sediments.

2.1.2.3 Types of Data Required--Spiked sediments as well as reference or control sediments must be analyzed for total solids, grain size, and total and dissolved organic carbon. The concentrations of chemicals added to sediment must be determined in stock solutions as well as in the test sediment. Bulk and interstitial concentrations of the spiked chemicals in the test sediment must be determined throughout a concentration range at least at the beginning and at the end of the toxicity test. However, methods for sampling interstitial water have not been standardized. If sediment properties that control availability, such as total volatile solids or metals, change during exposure, measurements must be taken before, during, and at the end of the exposure period, and the changes must be taken into account in interpreting the data. Sediment parameters such as pH and Eh should also be monitored.

Biological and chemical data are statistically compared with control or reference data to determine the occurrence of biological effects, and to calculate LC50 values, EC50 values, no-effect concentrations, or lowest-observable-effect concentrations. Establishment of the maximum acceptable toxicant concentration requires data from a chronic or life-cycle test.

Data correlating observed biological effects with chemical concentrations in spiked sediment can be used to calculate probit curves for derivation of biological effect level values (e.g., EC50, EC05). Data from several species of test organisms can be ranked, and the lowest contaminant concentrations that affect the most sensitive species can be used to establish sediment quality criteria that will protect the entire benthic community and associated aquatic ecosystem. This approach has regulatory and scientific precedence in the development of water quality criteria.

2.1.2.4 Necessary Hardware and Skills--Most toxicity test procedures require a minimum of specialized hardware and level of skill. In general, only readily available and inexpensive laboratory equipment is needed, procedures are fairly simple and straightforward, and a minimum of training is needed to detect and interpret biological endpoints. Chemical sampling methods are generally simple and routine, although analyses of chemical samples requires specialized training and equipment. Some biological effects tests also require specialized training and experience, especially to interpret the results.

2.1.3 Adequacy of Documentation--

Various acute sediment toxicity test procedures have been developed and are well documented for testing freshwater and marine field sediments (reviewed by Swartz 1987; Lamberson and Swartz 1988). Although only a few of these procedures have been used with laboratory-spiked sediments, most of the established methods could be used with laboratory-prepared sediments as well as with field sediments.

In contrast to acute tests, there are relatively few life-cycle test procedures for benthic invertebrates. Life cycle tests exist for the amphipod *Ampelisca abdita* (Scott and Redmond in press), the polychaetes *Neanthes arenaceodentata* (Pesch 1979) and *Capitella capitata* (Chapman and Fink 1984), and freshwater oligochaetes (Wiederholm et al. 1987). Chronic exposures to most sensitive life stages are also inherent in the benthic recolonization procedure (Hansen and Tagatz 1980).

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

Spiked-sediment toxicity tests incorporate a direct measure of sediment biological effects. This approach is the only method that can directly quantify the interactive effects of combinations of contaminants.

When chemical concentrations in tested biota are measured after a spiked-sediment toxicity test, uptake of contaminants by benthic organisms (i.e., bioaccumulation) can be predicted. As an important component of food webs in aquatic ecosystems, benthic organisms can contribute toxicants from contaminated sediments to higher levels of the aquatic food web and ultimately affect human health. Sediment quality criteria and bioaccumulation studies using sediment toxicity test methods can help to set limits on the disposal of toxic sediments and predict uptake of toxicants into food webs. Combined with chemical analysis of sediment samples and bulk sediment toxicity testing, these limits can be used to define areas where food species should not be consumed, or where direct contact with contaminated sediments can be hazardous to human health.

Bioaccumulation studies and sediment quality criteria established using data from spiked-sediment toxicity testing with several benthic species can also be used to protect benthic communities and aquatic species that feed upon the benthos. Assuming a sufficient mix of taxonomic groups, sediment quality criteria based on responses of the most sensitive species within a benthic community can be developed to protect the structure and function of the entire ecosystem (Hansen and Tagatz 1980).

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

The spiked-sediment toxicity test approach can be used to directly measure the effects of specific chemicals in various types of sediments in the laboratory and to establish unequivocal analysis of causal effects. Since test conditions are controlled, the method can be used to experimentally determine the effects of individual chemicals on mixtures of chemicals or aquatic biota (Plesha et al. 1988; Swartz et al. 1988, 1989), to establish pathways of toxicity, and to provide specific effects concentrations (e.g., LC50, EC50, no-effect concentration). The influence of various physical characteristics of the sediment on chemical toxicity can also be determined (Ott 1986; DeWitt et al. 1988). The available data represent concentrations at which toxicity occurs rather than numerical sediment quality criteria. However, U.S. EPA is currently conducting a research project that may provide insight into the minimum database necessary to establish a sediment quality criterion for fluoranthene based on toxicological data.

Concentration-response data have been generated using spiked-sediment toxicity test methods for a variety of chemicals, including both metals and organic compounds. Specific data are available for phenanthrene, fluoranthene, zinc, mercury, copper, cadmium, hexachlorobenzene, pentachlorophenol, Aroclors 1242 and 1254, chlordane, DDE, DDT, dieldrin, endosulfan, endrin, sevin, creosote, and kepone (Adams et al. 1985; Cairns et al. 1984; DeWitt et al. in press; Kemp and Swartz 1989; McLeese and Metcalfe 1980; McLeese et al. 1982; Swartz et al. 1986b, 1988, 1989; Tagatz et al. 1977, 1979, 1983; Word et al. 1987). Concentrations of non-ionic organic compounds are usually normalized to sediment organic carbon content. Normalizing factors for metals and other chemicals are currently under research.

3.0 USEFULNESS

3.1 Environmental Applicability

3.1.1 Suitability for Different Sediment Types--

The spiked-sediment toxicity test approach is suitable for any type of sediment. This approach can also be used to establish the bioavailable component of the sediment responsible for the observed toxicity. The effects of various physical properties of the sediment on chemical toxicity can be experimentally determined. In some cases, the physical or chemical properties of the test sediment such as salinity or grain size may limit the species that can be used for testing and a substitute species must be used (DeWitt et al. in press). In establishing sediment quality criteria, the effects of adverse physical or chemical properties of the sediment itself must be reflected. When factors controlling bioavailability (e.g., organic carbon) are known, data can be normalized to these factors and thus the approach can be applied to any sediment type.

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

A major advantage of the spiked-sediment toxicity test method is that it is suitable for all classes of chemicals. In addition, this is the only approach currently available that can empirically determine the interactive effects among individual chemicals in mixtures of contaminants such as are usually found in real-world sediments (Swartz et al. 1988a). This approach can be used to provide experimental validation of sediment quality criteria generated by other approaches.

3.1.3 Suitability for Predicting Effects on Different Organisms--

Theoretically, any organism can be used in spiked-sediment toxicity. To protect a biological community and to predict the effects of a toxicant on different organisms, test organisms should be selected on the basis of their sensitivity to contaminants, their ability to withstand laboratory handling, and their ability to survive in control treatments. In tests to determine the effects of toxicants on a particular biological community, the test species selected should be among the most sensitive species found in the community or comparably sensitive. If the most sensitive species are protected, the entire community should be protected.

3.1.4 Suitability for In-Place Pollutant Control--

Spiked-sediment toxicity testing can be used to develop sediment quality criteria that can then be used to determine the extent of the problem area, in monitoring temporal and spatial trends, and to assess the need for remedial action. Criteria can be used to set target cleanup levels and in post-cleanup monitoring of acceptable contaminant levels.

3.1.5 Suitability for Source Control--

When combined with wasteload allocation models, spiked-sediment toxicity tests can be used in source control to establish maximum allowable effluent concentrations or mass loadings of single chemicals and mixtures of chemicals.

3.1.6 Suitability for Disposal Applications--

Spiked-sediment toxicity tests can be used to predict biological effects of contaminants prior to approval of dredged material disposal or sewage outfall permits.

3.2 General Advantages and Limitations

3.2.1 Ease of Use--

Most sediment toxicity test procedures are simple to use, requiring limited expertise and standard inexpensive laboratory equipment. Only a few sublethal-effects tests require specialized training.

3.2.2 Relative Cost--

The cost of individual toxicity tests is relatively low because of the limited expertise and inexpensive equipment requirements (see Chapter 2, Bulk Sediment Toxicity Approach). Costs to implement this approach as a regulatory tool would be comparatively high, since it would require the collection of sediment chemistry data for comparison to data established by the sediment toxicity test method. The cost of developing a large toxicological database would be relatively high because of the large number of individual chemicals and sediments that would have to be tested. The cost of generating the chemical and toxicological data necessary to establish a sediment quality criterion for one chemical by this method is estimated to be $100,000.

3.2.3 Tendency to be Conservative--

Spiked-sediment toxicity tests, which are laboratory-controlled experiments, provide a high degree of accuracy and precision. They are sufficiently controlled to provide a true estimate of the toxicity of individual chemicals in sediment. Biological endpoints, species, and life stages of test organisms required for testing are analogous to water quality criteria minimum data requirements. Laboratory bioassays, especially acute toxicity tests, are inherently limited in their ability to reflect all of the ecological processes through which sediment contaminants may affect benthic ecosystems under field situations.

3.2.4 Level of Acceptance--

Spiked-sediment toxicity test methods, which follow the procedures and rationale used to develop water quality criteria, are easily interpreted, technically acceptable, and legally defensible. The procedures and resulting data have been accepted and published in peer-reviewed journal articles, and some procedures are in the process of standardization by the American Society of Testing and Materials (ASTM) subcommittee on sediment toxicology. A regulatory strategy through which data generated by these test methods to establish sediment quality criteria is under development.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

Spiked-sediment toxicity test methods are easily implemented by laboratories with typical equipment, requiring inexpensive glassware and little specialized training. Spiking sediments may require special handling facilities for preparing stock solutions of highly toxic substances, and the interpretation of some sublethal biological endpoints may require some degree of training and experience. In general, special expertise or elaborate facilities are not required for the biological tests, although analyses of chemical samples require special equipment and training, and quality control procedures are essential.

3.2.6 Level of Effort Required to Generate Results--

This procedure consists of a laboratory toxicity test, and requires a moderate effort to initiate and terminate an experiment. The data generated must be compiled and some calculations must be made to derive concentration-response relationships. The generation of chemical and biological data required for a large database of sediment quality values based on this approach would require a relatively large level of effort.

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

Sediment toxicity tests applied to spiked sediments provide an unequivocal analysis of cause-and-effect relationships between toxic chemicals and biological responses. Since the procedures follow the rationale used in development of water quality criteria, the methods are legally defensible. Toxicity tests have long been accepted by both the public and the scientific community as a basis for water quality criteria and dredged material testing.

3.2.8 Degree of Environmental Applicability--

The spiked-sediment toxicity test approach is applicable to a wide range of environmental conditions and sediment types. The confounding effects of sediment variables such as grain size and organic content can be either experimentally addressed using toxicity test methods, or addressed using normalization equations (DeWitt et al. 1988). A major advantage of the procedure is the ability to predict interactive effects of chemical mixtures such as would be found in field sediments.

3.2.9 Degree of Accuracy and Precision--

Since the sediment toxicity test is a laboratory-controlled experiment, results have a high degree of accuracy and precision. The procedure produces a direct dose-response data set for individual chemicals in sediment. Field validation of sediment criteria generated by this approach is required.

4.0 STATUS

4.1 Extent of Use

Spiked-sediment toxicity test procedures are under development in several laboratories. Spiking procedures, as well as biological test procedures, are in the process of standardization by ASTM's sediment toxicology subcommittee.

4.2 Extent to Which Approach Has Been Field-Validated

Spiked-sediment toxicity test values have not been well field-validated, although some results have been published (Plesha et al. 1988; Swartz et al. 1989). As more data and criteria values become available, they can be compared with existing information on contaminant concentrations in sediment in areas where biological effects have been observed. The effects of interactions among contaminants, as well as the effects of nonchemical sediment variables must be taken into consideration during field validation (Swartz et al. 1989).

4.3 Reasons for Limited Use

The approach is still in the developmental stage in several laboratories, and although some data have been generated and compared to field conditions, a relatively large effort will be needed to generate a large database. There have been few comparisons of methods and species sensitivity, and few chronic toxicity tests have been developed.

4.4 Outlook for Future Use and Amount of Development Yet Needed

The outlook for future use of sediment toxicity tests is promising where accurate, direct dose-response data are desired, or where the effects of chemical interactions need to be examined. Development of sediment spiking and biological testing methods should continue, methods should be compared and standardized among laboratories, and results should be field-validated to establish their ability to predict biological effects in sediments. As more toxicity tests are conducted, results should be compiled in a central database so that comparisons among species, methods, and laboratories can be made and sediment quality criteria can be developed.

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CHAPTER 4. INTERSTITIAL WATER TOXICITY APPROACH

Gerald Ankley and Nelson Thomas
U.S. Environmental Protection Agency
Environmental Research Laboratory
6201 Congdon Boulevard
Duluth, MN 55804
(218) 720-5702

The interstitial water toxicity approach is a multiphase procedure for assessing sediment toxicity using interstitial (i.e., pore) water. The use of pore water for sediment toxicity assessment is based on the strong correlations between contaminant concentrations in pore water and toxicity (and/or bioaccumulation) of sediment-associated contaminants by benthic macroinvertebrates (Adams et al. 1985; Swartz et al. 1985; Connell et al. 1988; DiToro 1988; Knezovich and Harrison 1988; Swartz et al. 1988; Giesy and Hoke in press). The approach combines the quantitation of pore water toxicity with toxicity identification evaluation (TIE) procedures to identify and quantify chemical components responsible for sediment toxicity (Mount and Anderson-Carnahan 1988a,b; Mount 1988). TIE involves recently developed techniques for the identification of toxic compounds in aqueous samples containing mixtures of chemicals. In the interstitial water toxicity method, TIE procedures are implemented in three phases to characterize pore water toxicity, identify the suspected toxicant, and confirm toxicant identification.

1.0 SPECIFIC APPLICATIONS

1.1 Current Use

The TIE procedures described herein were developed over the last 3 yr using municipal and industrial effluents from more than 30 sites. They have been used with several aquatic species including cladocerans and fishes, and they can be used with any type of benthic species that is amenable to toxicity testing in aqueous phases. Although the methods were developed largely with freshwater species, they are generally applicable to, and are currently being used with, marine organisms as well. The procedures have proven to be successful in identifying acutely toxic substances in more than 90 percent of the samples to which they have been applied. This success rate was achieved with a sample size of greater than 60 municipal and industrial effluents, surface water samples, and sediment fractions, including pore water and elutriates.

1.2 Potential Use

The use of pore water as a fraction to assess sediment toxicity, in conjunction with associated TIE procedures, can provide data concerning specific compounds responsible for toxicity in contaminated sediments. These data could be critical to the success of remediation of toxic sediments.

In spite of existing uncertainties in using pore water to assess sediment toxicity, the ability to identify specific toxicants responsible for acute toxicity in contaminated sediments makes pore water a potentially important sediment test fraction. Thus this method, in conjunction with other sediment classification methods, could prove to be extremely valuable.

2.0 DESCRIPTION

2.1 Description of Method

The interstitial water toxicity method involves three major steps:

- Isolation of pore water from sediment samples

- Performance of toxicity tests on pore waters

- Application of TIE procedures to pore water fractions.

Pore water can be isolated from sediment samples by compression (squeezing) techniques, displacement of water from sediment via the use of inert gases, centrifugation of bulk sediment, direct sampling of pore water through the use of dialysis membranes, and micro-syringe sampling (Knezovich et al. 1987; Knezovich and Harrison 1988; Sly 1988). The most representative pore water samples probably are obtained using the latter two procedures. However, the resulting sample volumes are too small to be useful for toxicity tests and associated TIE work. Centrifugation has been used in a number of studies evaluating the toxicity of sediment pore water (Giesy et al. 1988; Ankley et al. in press; Hoke et al. in preparation). However, there has been no critical evaluation of the relative advantages and disadvantages of the former three pore water preparation procedures in terms of toxicity assessment. Consequently, it would be premature to recommend one over another. With any of these pore water preparation techniques, care must be taken to avoid loss of contaminants due to oxidation, change in pH, or other interferences, during sample preparation.

After preparation of pore water, toxicity tests can be performed using either standard test species (U.S. EPA 1985a,b) or various types of benthic organisms amenable to toxicity testing in aqueous samples. In samples exhibiting acute toxicity, it is then possible to directly apply the TIE procedures described below in Section 2.1.2.2.

2.1.1 Objectives and Assumptions--

The objective of this approach is to derive toxicity data in the laboratory that can be used to assess sediment toxicity in field situations. With the interstitial water toxicity method, it is possible to quantify toxicity in a sample and potentially to identify chemical components responsible for toxicity. The major assumption in using this method is that the compounds that are toxic to test organisms in the pore water are the same compounds that cause toxicity in sediments *in situ*.

2.1.2 Level of Effort--

Implementation of this method requires a moderate amount of laboratory effort, both to perform toxicity tests and to conduct TIE studies. Effort expended in the TIE studies will be proportional to the complexity of analyses required for the identification of suspected toxicants.

2.1.2.1 Type of Sampling Required--Bulk sediment must be obtained and pore water prepared from the sediments. Routine measurement of certain chemical components of the pore water should be conducted. These measurements should include (but are not limited to) pH, hardness, alkalinity, salinity (where appropriate), dissolved oxygen, sulfides, and ammonia. Certain of these variables, in particular pH, also should be monitored in the bulk sediment.

2.1.2.2 Methods--The framework for existing TIE procedures is summarized below. Greater detail (e.g., with respect to all possible results that could be generated) is available in Mount and Anderson-Carnahan (1988a,b) and Mount (1988).

Toxic sediment samples can potentially contain thousands of chemicals, and usually only a handful are responsible for the observed toxicity. The goal of the TIE method is to identify quickly and cheaply the chemicals causing toxicity. However, components causing toxicity can vary widely and potential toxicants include cationic metals, polar and nonpolar organics, and anionic inorganics, as well as ammonia. In addition, when multiple toxicants are present, it must be possible to determine the proportion of the overall toxicity due to each toxicant.

After preparation of pore water and performance of initial toxicity tests, the initial step in the TIE process is to separate toxicants from nontoxic components in the pore water sample. To isolate the toxicants, sample manipulations and subsequent fractionation techniques are used in combination with toxicity tests (toxicity tracking). This approach allows the physical and chemical nature of the toxicants to be determined prior to instrumental analysis. Consequently, the "correct" analytical technique can be selected for detecting as well as identifying the toxicants in the subsample. In addition, significantly fewer chemical components are in the subsamples as compared to the original sample, and thus, the task of deciding which component is causing the toxicity is much easier. The toxicity-based TIE approach enables direct relationships to be established between toxicants and measured analytical data because toxicants are tracked through all sample fractionations, using the most relevant detector available, the organism. Establishing this relationship ultimately results in highly efficient TIEs.

With the toxicity-based TIE approach, detection of synergistic and antagonistic interactions, as well as matrix effects, for the toxicants is possible via toxicity tracking. *A priori* knowledge of the toxicants' behavior in the aqueous phase is not required.

The TIE approach is divided into three phases. Phase I consists of methods to identify the physical/chemical nature of the constituents causing acute toxicity. Phase II describes fractionation schemes and analytical methods to identify the toxicants, and Phase III presents procedures to confirm that the suspected toxicants are the cause of toxicity.

Phase I: Toxicant Characterization--In Phase I, the physical/chemical properties of toxicants are characterized by performing manipulations to alter or render biologically unavailable generic classes of compounds with similar properties. Toxicity tests, performed in conjunction with the manipulations, provide information on the nature of the toxicants.

Successful completion of Phase I occurs when both the nature of the components causing toxicity, as well as their consistent presence in a number of samples, can be established. After Phase I, the toxicants can be tentatively categorized as having chemical characteristics of cationic metals, nonpolar organics, polar organics, volatiles, oxidants, and/or substances whose toxicity is pH dependent.

An overview of the sample manipulations employed in Phase I is shown in Figure 4-1. Not shown in Figure 4-1, but performed on all samples are routine water chemistry measurements including pH, hardness, conductivity, and dissolved oxygen. These routine measurements are needed for designing sample manipulations, and interpreting test data. The manipulations shown in Figure 4-1 are usually sufficient to characterize toxicity caused by a single chemical. When multiple toxicants are present, various combinations of the Phase I manipulations will most likely be required for toxicant characterization.

Many of the manipulations in Phase I require samples that have been pH-adjusted. The adjustment of pH is a powerful tool for detecting cationic and anionic toxicants, since their behavior is strongly influenced by pH. By changing pH, the ratio of ionized to un-ionized species in solution for a chemical is changed significantly. The ionized and un-ionized species have different physical/chemical properties as well as toxicities. In Phase I, pH manipulations are used to examine two different questions:

- Is the toxicity different at various pHs?
- Does changing the pH, performing a sample manipulation, and then readjusting to ambient pH affect toxicity?

The graduated pH test examines the first question, and the pH adjustment, aeration, filtration, and solid phase extraction (SPE) manipulations examine the second.

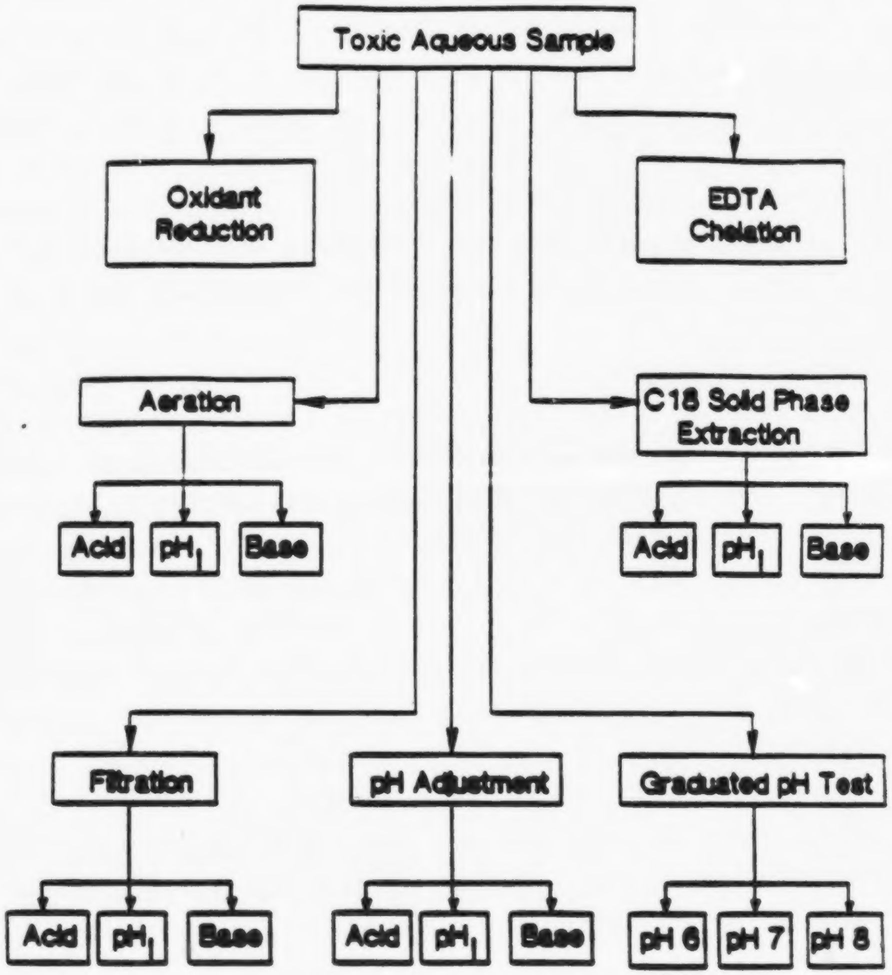
Toxic Aqueous Sample
Oxidant Reduction
EDTA Chelation
Aeration
C18 Solid Phase Extraction
Acid
pH$_I$
Base
Acid
pH$_I$
Base
Filtration
pH Adjustment
Graduated pH Test
Acid
pH$_I$
Base
Acid
pH$_I$
Base
pH 6
pH 7
pH 8

Figure 4-1. Overview of the Phase I toxicity characterization process. The ambient pH of the sample is indicated as pH$_I$.

In the graduated pH test, the pH of a sample is adjusted within a physiologically tolerable range (e.g., pH 6.0, 7.0, and 8.0) before toxicity testing. Generally, the un-ionized form of a toxicant is able to cross biological membranes more readily than the ionized form and thus, is more toxic. This test is designed primarily for ammonia, a relatively common toxicant whose toxicity is extremely pH-dependent (U.S. EPA 1985c). However, different pH values can strongly affect the toxicity of many common ionizable pesticides, and also may influence the bioavailability and toxicity of certain heavy metals (Campbell and Stokes 1985; Doe et al. 1988).

Aeration tests are designed to determine whether or not toxicity is attributable to volatile or oxidizable compounds. Samples at pH_I (ambient pH), pH 3, and pH 11 are sparged with air for 1 h, readjusted to pH_I, and tested for toxicity. The different pH values affect the ionization state of polar toxicants, thus making them more or less volatile, and also affect the redox potential of the system. If toxicity is reduced by air sparging at any of the pH values, the presence of volatile or oxidizable compounds is suggested. To distinguish the former from the latter situation, further experiments are performed using nitrogen rather than air to sparge the samples. If toxicity remains the same, oxidizable materials are implicated; if toxicity is again reduced, volatile compounds are suspect. The pH at which toxicity is reduced is important. If nitrogen sparging decreases toxicity at pH_I, neutral volatiles are present, whereas, if toxicity decreases at pH 11.0 or pH 3.0, basic and acidic volatiles, respectively, are implicated.

Filtration provides information concerning the amount of toxicity associated with filterable components. In this test, samples at pH_I, pH 3.0, and pH 11.0 are passed through 1-um filters, readjusted to pH_I, and tested for toxicity. Reductions in toxicity due to filtration could be related to factors such as decreased physical toxicity, rather than chemical toxicity (Chapman et al. 1987), or removal of particle-bound toxicants, which could be important, particularly if filter-feeding organisms such as cladocerans are the test species.

Reversed phase, solid phase extraction (SPE) is designed to determine the extent of toxicity due to compounds that are relatively nonpolar at pH_I, pH 3.0, or pH 9.0. This test, in conjunction with associated Phase II analytical procedures, is an extremely powerful TIE tool. In this procedure, filtered sample aliquots at pH_I, pH 3.0, and pH 9.0 are passed through small columns packed with an octadecyl (C_{18}) sorbent. At pH_I, the C_{18} sorbent will remove neutral compounds such as certain pesticides (Junk and Richard 1988). By shifting ionization equilibria at the low and high pH values, the SPE column also can be used to extract organic acids and bases (Wells and Michael 1987). During extraction, the resulting post-column effluent is collected and tested for toxicity in order to determine if the manipulation removed toxicity and/or if the capacity of the column was exceeded. If sample toxicity is decreased, a nonpolar toxicant would be suspected.

The presence of toxicity due to cationic metals is tested through additions of ethylenediaminetetraacetic acid (EDTA), a strong chelating agent that produces nontoxic complexes with many metals. As with SPE chromatography, the specificity of the EDTA test for a class of ubiquitous toxicants makes it a powerful TIE tool. Cations chelated by EDTA include certain forms of aluminum, barium, cadmium, cobalt, copper, iron, lead, manganese, nickel, strontium, and zinc (Stumm and Morgan 1981). EDTA does not complex anionic forms of metals, and only weakly chelates certain cationic metals (e.g., silver, chromium, thallium) (Stumm and Morgan 1981). Because EDTA nonspecifically binds a variety of cations, the appropriate range of EDTA concentrations to use in the test is highly dependent upon calcium and magnesium concentration (hardness) and salinity, as well as the sensitivity of the test organism to EDTA.

The oxidant reduction test is designed to determine the degree of toxicity associated with chemicals reduced by sodium thiosulfate. The toxicity of compounds such as chlorine, bromine, iodine, and manganous ions are neutralized by this treatment. Because sodium thiosulfate, like EDTA, has low toxicity to most aquatic organisms, a relatively wide range of concentrations can be tested.

Phase II: Toxicant Identification--Initial laboratory work in Phase II focuses on isolation of the toxicants using chemical fractionation techniques with toxicity tracking. The ideal isolation process would create a subsample that contains one chemical, the toxicant. Depending upon the nature of the toxicants, wide differences in the techniques as well as in the amount of effort required for fractionation will occur.

In general, after fractionation, instrumental analyses are performed on the toxic subsamples, and lists of identified chemicals are assembled for each subsample. For each chemical in a list, an LC50 value is obtained, usually from the literature or occasionally from structure activity models (Institute for Biological and Chemical Process Analyses 1986). By comparing concentrations of the identified chemicals to their LC50 values, a list of suspect toxicants is made. This list is then refined by actually determining LC50 values for the suspects using the TIE test species. If only one toxicant is present, it should be easily identified. For samples with multiple toxicants, identification becomes significantly more protracted, since interactions between toxicants may need to be examined. If none of the suspected toxicants appears to explain the toxicity, the true toxicants were probably not detected during instrumental analysis. Usually, additional separation, combined with concentration steps is required to increase the analytical sensitivity for toxicant identification.

The information obtained in Phase I provides the analytical roadmarks for performing the fractionation and identification tasks in Phase II. To illustrate the relationship between Phase I data and analytical approaches employed in Phase II, results for two typical Phase I TIE evaluations are presented in Table 4-1. The Phase II methods and approaches appropriate for these examples are discussed below.

In the first sample in Table 4-1, SPE reduced toxicity. In Phase II, the SPE column is eluted with graded, increasingly nonpolar methanol/water solutions, and toxicity testing is performed on each fraction. Although solvents other than methanol are routinely used in analytical work with C_{18} chromatography columns, the low toxicity of methanol to aquatic organisms (e.g., LC50 $\geq$1.5 percent for cladocerans) makes it a solvent of choice for toxicity tracking in the fractions. If no toxicity occurs in the fractions, the toxicants have been lost and further characterization (Phase I) work is required. If toxicity occurs in the fractions, Phase II methods feature concentration of the toxic methanol/water fractions, high performance liquid chromatography fractionation of the concentrate (again with a C_{18}/methanol/water solvent system) with concurrent toxicity testing of the fractions, and ultimately, identification of suspected toxicants in the toxic fractions via gas chromatography/mass spectroscopy.

In the second sample, both EDTA additions and SPE reduced toxicity. The reduction of toxicity by EDTA strongly suggests the presence of toxic levels of cationic metals. Thus, Phase II procedures would include both metal analyses and the concentration, fractionation, and identification techniques described for nonpolar organics in the first example. If analyses identify specific metals at concentrations high enough to cause toxicity, various mass balance procedures can be used to define the portion of the sample toxicity due to the suspected metals, and the portion of the toxicity due to the suspect nonpolar compounds.

Only a very small subset of possible Phase I results is shown in Table 4-1, particularly when one considers that three of the tests (aeration, filtration, SPE) are conducted at three different pH values. A complete discussion of the types of Phase I results that may be encountered and

TABLE 4-1. PHASE I CHARACTERIZATION RESULTS AND SUSPECT TOXICANT CLASSIFICATION FOR TWO EFFLUENTS

	Effluent[a]	
	One	Two
Phase I Test		
Oxidant reduction	NR	NR
EDTA addition	NR	R
Graduated pH test	NR	NR
pH adjustment	NR	NR
Filtration	NR	NR
Aeration	NR	NR
SPE	R	R
Suspected toxicant classification	Nonpolar organics	Nonpolar organics/ cationic metals

[a] NR = No reduction in toxicity.
R = Reduction in toxicity.

subsequent Phase II strategies that could be implemented is beyond the scope of this review.

Phase III: Toxicant Confirmation--After Phase II identification procedures implicate suspected toxicants, Phase III is initiated to confirm that the suspects are indeed the true toxicants. Confirmation is perhaps the most critical step of the TIE because procedures used in Phases I and II may create artifacts that could lead to erroneous conclusions about the toxicants. Furthermore, there is a possibility that substances causing toxicity are different from sample to sample within a supposedly homogeneous geographic region. Phase III enables both situations to be addressed. The tools used in Phase III include correlation, relative species sensitivity, observation of symptoms, spiking, and mass balance techniques. In most instances, no single Phase III test is adequate to confirm suspects as the true toxicants; it is necessary to use multiple confirmation procedures.

In the correlation approach, observed toxicity is regressed against expected toxicity due to measured concentrations of the suspected toxicants in samples collected over time or from several sites within a location. For the correlation approach to succeed, temporal or spatial variation has to be wide enough to provide a range of values adequate for meaningful analyses. In order to use the correlation approach effectively when there are multiple suspected toxicants, it is necessary to generate data concerning the additive, antagonistic, and synergistic effects of the toxicants in ratios similar to those found in the samples. These data also are needed for the spiking and mass balance techniques described below.

The relative sensitivity of different test species can be used to evaluate suspected toxicants. If there are two or more species that exhibit markedly different sensitivities to a suspected toxicant in standard reference tests, and the same patterns in sensitivity are seen with the toxic pore water sample, this provides evidence for the validity of the suspect being the true toxicant.

Another Phase III procedure is observation of symptoms (e.g., time to mortality) in poisoned animals. Although this approach does not necessarily provide support for a given suspect, it can be used to provide evidence against a suspected toxicant. If the symptoms observed in a standard reference test with a suspected toxicant differ greatly from those observed with the pore water sample (which contains similar concentrations of the suspected toxicant), this is strong evidence for a misidentification.

Confirmatory evidence can also be obtained by spiking samples with the suspected toxicants. While the results may not be conclusive, an increase in toxicity by the same proportion as the increase in concentration of the suspected toxicant in the sample suggests that the suspect is correct. To get a proportional increase in toxicity from the addition of a suspected toxicant when in fact it is not the true toxicant, both the true and suspected toxicants would have to have very similar toxicity levels and their effects would also have to be additive.

Mass balance calculations can be used as confirmation steps when toxicity can be at least partially removed from the pore water sample, and subsequently recovered. This approach can be useful in instances when SPE removes toxicity. The methanol fractions eluted from the SPE column are evaluated individually for toxicity; these toxicities are summed and then compared to the total amount of toxicity lost from the sample.

Other techniques, including alteration of water quality characteristics (e.g., pH, salinity) in a manner designed to affect the toxicity of specific compounds, and analysis of body burdens of suspected toxicants in exposed animals also can be useful confirmation steps.

2.1.2.3 Types of Data Required--In addition to the routine measurements described above, biological response data, either acute or chronic, will be obtained. Specific data collected will depend upon the choice of test organism. If the TIE process is initiated, the researcher will first obtain data concerning the physical/chemical characteristics of the toxicants in the pore water, followed by actual identification of toxic compounds, and standard determination of their concentrations in the toxic samples (see Section 2.1.2.2 above).

2.1.2.4 Necessary Hardware and Skills--Pore water preparation and toxicity test procedures are fairly straightforward, and require commonly available equipment and facilities. Many of the TIE procedures also require only routine facilities. However, certain TIE techniques require some degree of advanced analytical capability (e.g., atomic absorption spectroscopy, gas chromatography/mass spectroscopy). Similarly, although many of the routine toxicity tests require relatively little training, certain of the TIE procedures, in particular some of the chemical analyses, require an advanced degree of technical expertise and experience.

2.1.3 Adequacy of Documentation--

The theoretical basis for using pore water to assess toxicity appears to be scientifically sound, and thus, has been recommended for sediment toxicity evaluation (Adams et al. 1985; Swartz et al. 1985; Knezovich and Harrison 1988; Swartz et al. 1988; Connell et al. 1988; DiToro 1988; Giesy and Hoke in press). Toxicity tests that can be used are well-documented, standard procedures (U.S. EPA 1985a,b). The TIE techniques involved have been documented and evaluated (Mount and Anderson-Carnahan 1988a,b; Mount 1988). Also, sediment toxicity assessment with pore water, including toxicant identification, has been successfully demonstrated (Ankley et al. in press).

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

This method can be used to predict biological effects of toxic sediment on aquatic organisms, and can identify toxicants responsible for observed effects. The data generated thus can be used to design sediment remediation programs that would have an optimal likelihood of success. These procedures are not suitable for evaluating human health effects or protecting wildlife.

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

Pore water toxicity assessment, in conjunction with successful TIE procedures, can be used to generate numerical criteria for toxic compounds in sediment pore water, because the toxicants are actually identified. However, it must be established that compounds identified as being toxic to test organisms in the laboratory are the same compounds (both in form and concentration) responsible for toxicity to organisms in field situations. This relationship can be evaluated both through biosurveys (possibly in conjunction with analysis of contaminant residues in organisms collected from the field), and laboratory toxicity tests in which benthic organisms perceived to be impacted in contaminated sediments *in situ* are exposed to toxicants identified in the pore water. Both types of data also would be required for any sediment classification method based on toxicity or chemical analyses.

3.0 USEFULNESS

3.1 Environmental Applicability

3.1.1 Suitability for Different Sediment Types--

The pore water toxicity assessment approach is suitable for any sediment from which adequate quantities of pore water can be isolated. In typical sediments, 20-50 percent of the volume of the bulk sediment sample is pore water. For a complete Phase I characterization with a test species of relatively small body size (e.g., cladocerans, larval fishes), approximately 1.5 L of pore water is required. This translates into a bulk sediment requirement of 3-8 L. Bulk sediment volumes needed for Phase II identification will, of course, be dependent upon the toxicants present in the pore water, but typical volumes required would be expected to range from 1 to 20 L.

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

This approach appears to be suitable for various water soluble nonpolar organics, cationic metals, and ammonia (Adams et al. 1985; Swartz et al. 1985; Knezovich and Harrison 1988; Swartz et al. 1988; Connell et al. 1988; DiToro 1988; Ankley et al. in press). The applicability of the approach to toxicants such as polar organics or extremely lipophilic compounds has yet to be established. Also, the TIE procedures enable the evaluation of interactive (additive, synergistic, antagonistic) effects among various toxicants present in pore water samples (Mount and Anderson-Carnahan 1988a,b; Mount 1988).

3.1.3 Suitability for Predicting Effects on Different Organisms--

If the TIE procedures successfully identify specific toxicants responsible for sediment toxicity, the impacts of these toxicants on various species of concern can be easily predicted, provided that there are data concerning the toxicity of the identified compounds to these species. Although toxicity data may not be available for certain benthic species, once suspected toxicants are identified, it would be possible to generate toxicity data for specific species of concern.

3.1.4 Suitability for In-Place Pollutant Control--

The pore water toxicity assessment method and associated TIE procedures could prove to be a powerful tool for in-place pollutant control. Because sediment toxicants are actually identified, it is possible to design remediation plans for toxicants from point sources or controllable nonpoint sources, and to routinely monitor the success of these plans through continued assessment of pore water for toxicity and specific chemical toxicants.

3.1.5 Suitability for Source Control--

Because the potential exists for identifying specific sediment toxicants, this method is ideal for point source control, as well as controllable nonpoint sources.

3.1.6 Suitability for Disposal Applications--

As stated above, because specific sediment toxicants can be identified, it would be possible to identify potential hazards of contaminated sediments to aquatic organisms before disposal operations.

3.2 General Advantages and Limitations

3.2.1 Ease of Use--

Pore water preparation, routine chemical analyses, toxicity tests, and certain of the TIE procedures are reasonably straightforward and require relatively little technical expertise or extensive laboratory facilities. Because it is possible to work with aqueous samples, many of the standard toxicity tests developed for toxicity assessment of surface waters and effluents can be utilized, in addition to tests with various benthic species (U.S. EPA 1985a,b). However, interpretation of results of certain of the TIE procedures, as well as analytical support for the TIE work, requires advanced training and experience. At present, there are no set protocols for the preparation of pore water, and there is uncertainty about changes in pore water chemistry after extraction. Also, several TIE analyses require highly sensitive analytical instrumentation for procedures, such as atomic absorption spectroscopy and gas chromatography/mass spectroscopy.

3.2.2 Relative Cost--

Cost of the actual toxicity test procedures is relatively low. Cost of the TIE procedures will vary depending upon the nature of the toxic compounds; certain toxicants (e.g., pesticides) are more costly to identify and quantify than others (e.g., ammonia). Also, identification and determination of the effects of multiple toxicants in samples cost more than the identification of single toxicants. Thus, cost analysis for the TIE portion of the toxicity assessment is case-specific.

3.2.3 Tendency to be Conservative--

Depending upon the species used and the endpoint evaluated, pore water toxicity tests can be as conservative as desired.

3.2.4 Level of Acceptance--

The theoretical basis of pore water toxicity assessment is sound (Adams et al. 1985; Swartz et al. 1985; Knezovich and Harrison 1988; Swartz et al. 1988; Connell et al. 1988; DiToro 1988; Giesy and Hoke in press). The most important advantage of utilizing pore water as a sediment test fraction, however, is the fact that it enables the application of recently developed TIE procedures for the identification of toxic compounds in aqueous samples containing complex mixtures of chemicals (Mount and Anderson-Carnahan 1988a,b). These procedures are not available for direct chemical analyses of sediments. TIE procedures have proven to be extremely powerful tools for work with complex effluents, and can be used with any type of acutely toxic aqueous sample, including sediment pore water (Ankley et al. in press). The ability to identify specific compounds responsible for toxicity of contaminated sediments clearly could be critical to the success of remediation.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

Pore water preparation, toxicity test procedures, and certain of the TIE methods are easily implemented by laboratories with typical equipment and a moderate degree of expertise. Interpretation of some TIE results requires additional technical training and experience, and certain of the analytical procedures associated with TIE work require both specialized training and analytical instrumentation.

3.2.6 Level of Effort Required to Generate Results--

This procedure consists of field sampling, preparation of pore water, toxicity tests, and various TIE procedures. Depending upon the results of the TIE work, the level of effort expended to obtain potentially important data can be relatively small.

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

Biological responses (i.e., toxicity) can be easily interpreted, and when properly performed, the results of the TIE procedures are straightforward and easily interpreted by personnel with appropriate backgrounds.

3.2.8 Degree of Environmental Applicability--

Pore water toxicity assessment and TIE procedures are applicable to virtually all environmental conditions and sediment types. Moreover, a wide variety of test organisms can be evaluated with this approach. However, although data indicate that the toxicity and/or bioaccumulation of a variety of contaminants is correlated with their pore water concentrations, there is no guarantee that this relationship exists for all types of contaminants. For example, a potentially important route of exposure for highly lipophilic compounds is thought to be via ingestion of contaminated particles. This route is not addressed using pore water exposures. Finally, existing TIE procedures are applicable for acutely toxic samples, and thus generally would not be useful for identifying chronically toxic sediment contaminants.

3.2.9 Degree of Accuracy and Precision--

Because the procedures consist of laboratory controlled experiments, results obtained are statistically accurate and precise.

4.0 STATUS

4.1 Extent of Use

Various toxicity tests have been widely applied to the evaluation of both freshwater and marine sediments, and pore water is merely one of the possible fractions that can be tested. Theoretically, pore water appears to be appropriate for sediment toxicity assessment (Adams et al. 1985; Swartz et al. 1985; Knezovich and Harrison 1988; Swartz et al. 1988; Connell et al. 1988), and thus, it has been recommended as a suitable fraction for the evaluation of sediment toxicity (DiToro 1988; Giesy and Hoke in press). The TIE procedures (Mount and Anderson-Carnahan 1988a,b; Mount 1988), although developed only relatively recently, already are widely used both in research and regulatory programs.

4.2 Extent to Which Approach Has Been Field-Validated

Because the procedure is very new, there has been little field validation. This area requires research, not only for the pore water method described herein, but for virtually any other sediment classification method involving toxicity tests or chemical analyses.

4.3 Reasons for Limited Use

Various sediment toxicity tests have been widely used; however, relatively few studies have evaluated pore water toxicity. This is primarily because the theoretical basis for utilizing pore water has only recently been critically evaluated. For this reason, there are no standard methods for pore water preparation. Systematic TIE procedures for toxic aqueous samples have only recently been developed, and thus, have not yet been widely applied to the area of sediment toxicity assessment. Because current TIE procedures cannot be used with bulk sediment samples, pore water appears to be the best fraction with which to attempt to identify specific sediment contaminants responsible for acute toxicity.

4.4 Outlook for Future Use and Amount of Development Yet Needed

The outlook for this approach is extremely promising, because it is the only method currently available which enables the identification of specific compounds responsible for sediment toxicity with some degree of certainty.

This information could be critical to the success of remediation. However, as with all of the existing sediment classification methods, further development is needed, particularly in the following areas:

- Development of standard and scientifically sound techniques for pore water isolation
- Further characterization of relationships between sediment toxicity *in situ* and the toxicity of sediment pore water in the laboratory for different classes of compounds
- The development of TIE procedures to identify chronically toxic compounds in aqueous samples (research in this area is ongoing at ERL-Duluth, primarily with complex effluents).

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CHAPTER 5. EQUILIBRIUM PARTITIONING APPROACH

Christopher S. Zarba
U.S. Environmental Protection Agency
401 M Street S.W. (WH-585)
Washington, DC 20460
(202) 475-7326

The equilibrium partitioning (EP) approach focuses on predicting the chemical interaction among sediments, interstitial water (i.e., the water between sediment particles), and contaminants. Based on correlations with toxicity, interstitial water concentrations of contaminants appear to be better predictors of biological effects than do bulk sediment concentrations. The EP method for generating sediment quality criteria is based on predicted contaminant concentrations in interstitial water vs. chronic water quality criteria. Chemically contaminated sediments are expected to cause adverse biological effects if the predicted interstitial water concentration for a given contaminant exceeds the chronic water quality criterion for that contaminant.

1.0 SPECIFIC APPLICATIONS

Specific applications of EP-based sediment quality criteria are under development. The primary use of EP-based sediment criteria will be to identify risks associated with contaminants. Because the regulatory needs vary widely among and within U.S. EPA offices and programs, EP-based sediment quality criteria may be used in many different ways.

EP-based numerical sediment quality criteria would likely be used directly to assess risk and applied in a tiered approach. In tiered applications, concentrations of sediment contaminants that exceed sediment quality criteria would be considered as causing unacceptable impacts. Further testing may or may not be required, depending on site-specific conditions. Sediment contaminants at concentrations less than the sediment criteria would not be of concern. However, sediments would not be considered safe in cases where they are suspected to contain other contaminants at concentrations above safe levels, but for which no sediment criteria exist. Synergistic, antagonistic, or additive effects of multiple contaminants in the sediments may also be of concern. Additional testing in other tiers of the evaluation approach, such as bioassays, could be required to determine whether the sediment is safe. It is likely that such testing would incorporate site-specific considerations.

1.1 Current Use

Specific regulatory uses of EP-based sediment quality criteria are under development. The method is presently being reviewed by the U.S. EPA Science Advisory Board to determine its suitability for generating sediment criteria for non-ionic contaminants. This review should be completed prior to establishing any formal framework for the application of sediment criteria. (The EP approach was presented to the Science Advisory Board on 2 February 1989. Their report is expected in July 1989.) The range of potential applications of the EP approach is large because the approach accounts for contaminant bioavailability and can be used to evaluate most sediments.

Interim sediment criteria values have been developed for a variety of organic compounds using the EP approach. In a pilot study at six Superfund sites at which site characterization and evaluation activities were undertaken, the interim criteria were used in the following ways:

- Identify the extent of contamination

- Assess the risks or potential risks associated with the sediment contamination
- Identify the environmental benefit associated with a variety of remedial options.

In addition, the State of New York has used interim EP-based sediment criteria to evaluate the potential effects of sediment contaminants found in aquatic habitats in that state.

1.2 Potential Use

Potential applications of the EP approach include a variety of ongoing activities by the U.S. EPA. EP-based sediment quality criteria could play a major role in the identification, monitoring, and cleanup of contaminated sediment sites on a national basis. They could also be used to ensure that uncontaminated sites remain uncontaminated. In some cases, such sediment criteria alone will be sufficient to identify and establish cleanup levels for contaminated sediments. In other cases, it will be necessary to supplement the sediment criteria with biological sampling, testing, or other types of analysis before a decision can be made.

EP-based sediment criteria will be particularly valuable at sites where sediment contaminant concentrations are gradually increasing. In such cases, criteria will permit an assessment of the extent to which unacceptable contaminant concentrations are being approached, or have been exceeded. Comparisons of field measurements to sediment criteria will be a reliable method for providing an early warning of a potential problem. Such an early warning would provide an opportunity to take corrective action before adverse impacts occur.

Although sediment criteria developed using the EP approach are similar in many ways to existing water quality criteria, their applications may differ substantially. In most cases, contaminants in the water column need only be controlled at the source to eliminate unacceptable adverse impacts. In contrast, contaminated sediments often have been in place for quite some time, and controlling the source of that pollution (if the source still exists) will not be sufficient to alleviate the problem. Safe removal, treatment, or disposal of contaminated sediments can also be difficult and expensive. For this reason, it is anticipated that EP-based sediment criteria will rarely be used as mandatory cleanup levels. Rather, they will be used to predict or identify the degree and spatial extent of problems associated with contaminated areas, and thereby facilitate regulatory decisions.

2.0 DESCRIPTION

2.1 Description of Method

Concentrations of contaminants in the interstitial water (i.e., the water between the sediment particles) correlate very closely with toxicity, whereas concentrations of contaminants bound to the sediment particles do not. The EP method for generating sediment criteria involves predicting contaminant concentrations in the interstitial water, and comparing those concentrations to quality criteria. If the predicted sediment interstitial water concentration for a given contaminant exceeds its respective chronic water quality criterion, then the sediment would be expected to cause adverse effects.

The processes that govern the partitioning of chemical contaminants among sediments, interstitial water, and biota are better understood for some kinds of chemicals than for others. Concentrations of manganese oxide, iron oxide, iron sulfide, and organic carbon are the primary factors that control phase associations, and therefore bioavailability, of trace metals in sediments. However, models that can use these factors to predict research and are not fully developed. Mechanisms that control the partitioning of polar organic compounds are also poorly understood. However, polar organic contaminants are not generally considered to be a significant problem in sediments. Partitioning of non-ionic organic compounds between sediments and interstitial water is highly correlated with the organic carbon content of sediments. Also, the toxicity of non-ionic organic contaminants in sediments is highly dependent on their interstitial water concentrations. Consequently, to date, the EP approach is well developed for non-ionic organic contaminants and is in the process of development for trace metals.

Interstitial water concentrations can be calculated using partition coefficients for specific non-ionic organic chemicals and criteria continuous concentrations from WQC documents. The sediment quality criterion for a specific chemical is defined as the solid phase concentration that will result in an uncomplexed interstitial water concentration equal to the chronic water quality criterion for that chemical. The rationale for using water quality criteria as the effect concentrations for benthic organisms is that the sensitivity range for benthic organisms appears to be similar to the sensitivity range for water column organisms. Moreover, partition coefficients for a wide variety of contaminants are available.

The calculation procedure for non-ionic organic contaminants is as follows:

$$rSQC = Kp * cWQC$$

where:

cWQC = Criterion continuous concentration
rSQC = Sediment quality criterion (ug/kg sediment)
Kp = Partition coefficient for the chemical (L/kg sediment) between sediment and water.

The method for calculating sediment quality criteria using the EP approach for contaminants other than non-ionic organic contaminants is under development.

2.1.1 Objectives and Assumptions--

Three principal assumptions underlie use of the EP-based approach to establish sediment quality criteria:

- For sediment-dwelling organisms, the uncomplexed interstitial water concentration of a chemical correlates with observed biological effects across sediment types, and the concentration at which effects are observed is the same as that observed in a water-only exposure (see Document 18 in Section 5.0)

- Partitioning models permit calculation of uncomplexed interstitial water concentrations of the chemical phases of sediments controlling availability

- Benthic organisms exhibit a range of sensitivities to chemicals that is similar to the range of sensitivities exhibited by water column organisms (see Document 18 in Section 5.0).

Data exist supporting each of these assumptions.

2.1.2 Level of Effort--

2.1.2.1 Type of Sampling Required--Sufficient sediment chemistry sampling is required to adequately characterize the area of concern. Total organic carbon concentrations are also needed, preferably for each sampling station.

2.1.2.2 Types of Data Required--Analyses are needed to determine the concentrations of the contaminants of concern in the sediment (bulk sediment analysis), and the concentrations of organic carbon in the sediment.

2.1.2.3 Necessary Hardware and Skills--The investigator must be able to design an appropriate sampling study, conduct bulk sediment analyses, operate a pocket calculator, and understand developed values and what they protect.

2.1.3 Adequacy of Documentation--

The method is very well documented (see Section 5.0).

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

Sediment criteria can be protective of human health, aquatic life, and wildlife. At present, only interim sediment criteria values that are protective of aquatic life have been developed. EP-based sediment criteria are derived directly from water quality criteria. Sediment criteria derived using water quality criteria are designed to be similar in their levels of protection, and would be as protective of human health, aquatic life, and wildlife as are water quality criteria.

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

The EP method generates numerical criteria for specific chemicals. Interim sediment quality criteria have been developed for the following chemicals:

- PAH

    - Acenaphthene
    - Aniline
    - Phenanthrene

- Pesticides

    - Chlordane
    - Chlorpyrifos
    - DDT
    - Dieldrin
    - Endrin
    - Ethyl
    - Parathion
    - Heptachlor
    - Heptachlor epoxide
    - Gamma-hexachlorocyclohexane (lindane)

- PCBs.

Techniques for developing sediment criteria for metal contaminants are under development at U.S. EPA laboratories and by contractors.

3.0 USEFULNESS

3.1 Environmental Applicability

One of the principal reasons for selecting the EP approach is that it is applicable in a wide variety of aquatic systems, which is a prerequisite for the development of national sediment quality criteria.

3.1.1 Suitability for Different Sediment Types--

Although aspects of the EP method are still under development, it is expected that sediment criteria for non-ionic contaminants developed using this approach will be applicable to all types of sediments found in both freshwater and marine environments. Additional work is needed to clarify the best use of the EP approach for sediments with less than 0.5 percent organic carbon.

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

The EP method for developing sediment criteria has been modified for different types of contaminants. Non-ionic, ionic, and metal contaminants all interact with sediment particles in different ways, and partitioning models have to be modified to account for these differences. The technical approach for developing sediment criteria for non-ionic organic contaminants has been well developed and is under peer review. The technical approach for developing sediment criteria for metal contaminants is under development and is expected to undergo peer review in 1991. Ionic contaminants are not believed to cause major problems in sediments, but work plans for sediment criteria development methods for these compounds have been written.

3.1.3 Suitability for Predicting Effects on Different Organisms--

As indicated above (see Section 2.1), the EP approach is based on predicted interstitial water concentrations of non-ionic organic contaminants, and comparisons of these concentrations with chronic water quality criteria. Typically, water quality criteria are based on toxicity information (e.g., median lethal or median effective concentrations) for a wide number of species, and are set low enough to be protective of at least 95 percent of the species tested. Consequently, exposure levels that are predicted using the EP approach can be compared with a range of toxic effects values that are representative of the different kinds of organisms upon which water quality criteria are based.

3.1.4 Suitability for In-Place Pollutant Control--

The EP method is suitable for in-place pollution control because it can be used to identify locations where concentrations of individual contaminants are causing adverse effects. Target cleanup levels can be identified, and the success of cleanup activities can be determined.

3.1.5 Suitability for Source Control--

The EP method is suitable for source control. This method predicts the concentration of a contaminant above which adverse impacts are likely. A direct measure of biological effects is not needed to identify safe levels.

3.1.6 Suitability for Disposal Applications--

The EP method is suitable for predicting the effects that contaminated sediments may have if moved to an aquatic site. It is not applicable to contaminated sediments that are disposed of at upland sites.

3.2 General Advantages and Limitations

The EP approach offers the following advantages:

- It is consistent with existing water quality criteria
- It relates risks to specific substances and it can be used to identify probable species at risk
- It is applicable across all types of sediments and in all types of aquatic environments, including lentic, lotic, marine, and estuarine environments
- Only site-specific chemistry data are needed
- Site-specific or station-specific sediment criteria can be calculated as soon as sediment chemistry data are available
- It incorporates the large quantities of data that were used in the development of water quality criteria
- It can be incorporated into existing regulatory mechanisms with little or no need for additional staffing or skills
- The equilibrium partitioning theory upon which it is based is well developed

- It can be modified easily to accommodate site-specific circumstances
- It can be used to identify risks to humans and wildlife that may occur as a result of bioaccumulation
- It identifies the degree of sediment contamination, and permits an assessment of whether contaminant concentrations are approaching an effects level.

The EP approach is limited in the following ways:

- Sediment criteria developed using this approach do not address possible synergistic, antagonistic, or additive effects of contaminants
- Interim sediment criteria presently exist for only 12 contaminants
- The technical approach for developing sediment criteria for metal contaminants is still under development
- Sediment quality criteria for non-ionic chemicals apply to sediments that have an organic carbon concentration ≥ 0.5 percent
- Sufficient water-only toxicity data do not exist for all contaminants of concern.

3.2.1 Ease of Use--

The calculation of site-specific sediment criteria is relatively easy, provided that sediment chemistry data adequately characterizing the site and water quality criteria protective of the desired organism are available.

3.2.2 Relative Cost--

Because site-specific biological data are not needed, the costs associated with this method depend primarily on the cost of collecting site-specific chemistry data.

3.2.3 Tendency to be Conservative--

Sediment criteria are derived using the chronic water quality criteria as effect levels. Hence, the levels of protection afforded by sediment criteria are similar to those of water quality criteria. In general, water quality criteria are deemed to be protective of 95 percent of the organisms most of the time.

3.2.4 Level of Acceptance--

The EP approach and its use in deriving sediment quality criteria are the result of the efforts of many scientists who represent a variety of federal agencies, industries, environmental organizations, universities, U.S. EPA laboratories, state agencies, and other institutions. These scientists were involved with the selection of the EP approach for generating sediment criteria, and have also played a role in development of the method. Papers that discuss various aspects of this effort have been presented at scientific conferences.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

No special laboratory facilities or requirements are needed. Sediment chemistry analysis is all that is required.

3.2.6 Level of Effort Required to Generate Results--

The necessary level of effort varies substantially from site to site, and is dependent on many factors. Compared with other methods, the EP method generates results quickly and more cost-effectively. No site-specific biological data are required.

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

All sediment evaluation procedures require some level of interpretation. However, a sediment criterion that is bracketed with an appropriate degree of uncertainty can provide pertinent information. For example, sediment chemistry data that identify concentrations below the conservative effect level for a particular contaminant could be deemed safe for that contaminant. A contaminant concentration above the upper uncertainty level could be identified immediately as contaminated, and some degree of contamination could be assigned to those sediments for the individual contaminant. Sediments whose concentration of a particular contaminant fall within the degrees of uncertainty would require more detailed interpretation, and possibly additional testing.

3.2.8 Degree of Environmental Applicability--

EP-based sediment quality criteria can be applied directly to any contaminated sediment containing >0.5 percent ionic carbon and non-ionic chemicals for which criteria are available. Extensive data analysis and site-specific biological data are not required to use sediment criteria developed using this method. (In some cases these attributes may nonetheless be desirable.) As a result, the EP method can be considered environmentally applicable in some cases. Because a wide variety of contaminated sediment sites exist, absolute statements regarding environmental applicability are difficult to make. However, the EP method would be appropriate in many situations to predict bioavailability, bioaccumulation, and biological effects.

3.2.9 Degree of Accuracy and Precision--

Each sediment criterion value developed using the EP method will have an associated degree of uncertainty, which will vary from criterion to criterion. The principal uncertainties associated with sediment criteria developed using the EP method are those associated with partition coefficients. Hence, each developed sediment criterion should be bracketed with uncertainty, thereby providing decision-makers with a greater understanding of the meaning of the developed values.

4.0 STATUS

The method for developing sediment criteria for non-ionic organic contaminants has been developed and is currently being reviewed by the U.S. EPA Science Advisory Board. Final comments are expected by July 1989. Guidelines and guidance on the development and use of sediment criteria are in early stages of development. The method for developing sediment criteria for metal contaminants is being investigated and results are promising. The metals method is expected to be sufficiently well-developed for peer review by 1991.

4.1 Extent of Use

Specific regulatory uses for EP-based sediment quality criteria have not been established. A formal framework for the application of sediment criteria is not expected until the U.S. EPA Science Advisory Board completes its review. The range of potential applications is very large because the need for evaluating potentially contaminated sediments arises in many contexts.

Interim sediment criteria values were developed for a variety of organic compounds. These values were used in a pilot study at six Superfund sites where site characterization and evaluation activities were conducted. The interim criteria were used in three ways:

- To identify the extent of contamination
- To assess the risks associated with sediment contamination
- To identify the environmental benefits associated with a variety of remedial options.

The State of New York has also used interim sediment criteria to evaluate the potential effects of several contaminants found in sediments in state waters.

4.2 Extent to Which Approach Has Been Field-Validated

Field data were used to compare predicted effects with actual field effects. The comparison was conducted by developing Screening Level Concentration for various contaminants and organisms. A pilot field verification study is underway in Puget Sound, where field sediments are being used to conduct laboratory experiments. Additional field verification of this method is needed, and will be conducted in FY90.

4.3 Reasons for Limited Use

The EP method is not commonly used for the following reasons:

1. It has been developed only recently, and sufficient time has not elapsed for the principles to be understood and used by others
2. The U.S. EPA Science Advisory Board review of this method has not been completed
3. The U.S. EPA has not yet developed and issued guidance on the use of this method
4. The EP method has not yet been formally adopted by EPA.

4.4 Outlook for Future Use and Amount of Development Needed

This method is the only procedure for derivation of sediment quality criteria that is generic across sediments, accounts for bioavailability of chemicals, and relates effects to specific chemicals. Therefore, it is likely that EP-based sediment quality criteria will be used much as water quality criteria are being used to define environmentally acceptable concentrations. Sediment quality criteria along with sediment toxicity tests analogous to water quality criteria and whole effluent toxicity tests could play major role in U.S. EPA's regulations of contaminated sediment.

5.0 DOCUMENTS

0) Initial Evaluation of Alternatives for Development of Sediment Related Criteria for Toxic Contaminants in Marine Waters (Puget Sound) 10/83 Phase I: Development of Conceptual Framework Phase II: Development and Testing of the Sediment-Water Equilibrium Partitioning Approach

1) Background and Review Document on the Development of Sediment Criteria 6/85

2) Sediment Quality Criteria Development Workshop 2/85

3) National Perspective on Sediment Quality 7/85

4) Elaboration of Sediment Normalization Theory for Nonpolar Hydrophobic Organic Contaminants 1/86

5) Protocol for Sediment Toxicity Testing For Nonpolar Organic Compounds 2/86

6) An Activity-Based Model for Developing Sediment Criteria for Metals: I. A New Approach 6/86

7) Sediment Quality Criteria Validation: Calculation of Screening Level Concentrations from Field Data 7/86 Attachment: Recalculation of Screening Level Concentrations for Nonpolar Organic contaminants in Marine Sediments 12/87

8) Guidance for Sampling of and Analyzing for Organic Contaminants in Sediments 1/87

9) Sediment Quality Criteria for Metals: III Review of Data on Re-complexation of Trace Metals by Particulate Organic Carbon 1/87

10) Regulatory Applications of Sediment Criteria 6/87

11) Evaluation of the Equilibrium Partitioning Theory for Estimating the Toxicity of the Nonpolar Organic Compound DDT to the Sediment Dwelling Amphipod Rhepoxynius Abronius 8/87

12) Sediment Quality Criteria for Metals: IV Surface Complexation and Acidity Constants for Modeling Cadmium and Zinc Adsorption on to Iron Oxides 8/87

13) Sediment Quality Criteria for Metals: II Review of Methods for Quantitative Determination of Important Adsorbents and Sorbed Metals in Sediments 8/87

14) Sediment Quality Criteria Methodology Validation: Uncertainty Analysis of Sediment Normalization Theory for Nonpolar Organic Contaminants 11/87

15) Reconnaissance Field Study for Verification of Equilibrium Partitioning: Nonpolar Hydrophobic Organic Chemicals 11/87

16) Sediment Quality Criteria for Metals: V Optimization of Extraction Methods for Determining the Quantity of Sorbents and Adsorbed Metals in Sediments 12/87

17) Interim Sediment Criteria Values for Nonpolar Hydrophobic Organic Contaminants 5/88

18) Briefing Report to the EPA Science Advisory Board on the Equilibrium Partitioning Approach to Generating Sediment Quality Criteria 1/89

CHAPTER 6. TISSUE RESIDUE APPROACH

Anthony R. Carlson
U.S. Environmental Protection Agency,
Environmental Research Lab-Duluth
6201 Congdon Blvd
Duluth, MN 55804
(218) 720-5523
FTS 780-5523

Philip M. Cook
U.S. Environmental Protection Agency,
Environmental Research Lab-Duluth
6201 Congdon Blvd
Duluth, MN 55804
(218) 720-5553
FTS 780-5553

Henry Lee II
U.S. Environmental Protection Agency,
Environmental Research Lab-Newport
Marine Science Drive
Newport, OR 97365
FTS 867-4042

In the tissue residue approach, sediment chemical concentrations that will result in acceptable residues in exposed biotic tissues are determined. Concentrations of unacceptable tissue residues may be derived from toxicity tests performed during generation of chronic water quality criteria, from bioconcentration factors derived from the literature or generated by experimentation, or by comparison with human health risk criteria associated with consumption of contaminated aquatic organisms. The tissue residue approach generates numerical criteria and is most applicable for non-polar organic and organometallic compounds.

1.0 SPECIFIC APPLICATIONS

1.1 Current Use

Tissue residues of chemical contaminants in aquatic organisms, particularly fish, are frequently used as measures of water quality in both freshwater and marine systems. The tendency of organisms to bioaccumulate chemicals from water and food is one of the factors used in establishing national water quality criteria (WQC) for the protection of aquatic life (Stephan et al. 1985). Non-polar organic chemicals, which may bioaccumulate to levels that are toxic to organisms or render the organisms unfit for human food, generally will also be found as sediment contaminants. Hydrophobic organic chemicals preferentially distribute into organic carbon in sediment and lipid in aquatic biota. The tissue residue approach has been used recently to establish the amount of reduction of 2,3,7,8-TCDD concentration in Lake Ontario sediments that will result in attainment of acceptable TCDD levels in fish (Cook et al. 1989). The acceptable sediment TCDD concentration is being used as a sediment criterion to determine the remedial action necessary to reduce the incremental loading of TCDD from the Hyde Park Superfund site to Lake Ontario (Carey et al. 1989). Tissue residues of benthic organisms have also been used in some regulatory actions, such as the assessment of bioaccumulation potential of dredged materials.

1.2 Potential Use

Although tissue residues have been used more commonly to determine the potential for bioaccumulation of chemical contaminants from sediments and dredged materials, they also provide an excellent measure of "effective exposure dose" - a measure of an organism's actual exposure over time to a pollutant of concern. This exposure measure may be related to the dose expected at the water quality criterion or directly to the potential for producing chronic toxic effects. Given the ability to measure or predict tissue residues resulting from exposures in contaminated sediment systems, it is possible to establish sediment criteria based on residue-toxicity effects relationships. These relationships can provide a basis for sediment criteria that are free of uncertainties normally associated with organism exposures and sediment contaminant bioavailability. This is especially true when *in situ* measurements provide the basis for the sediment residue link to the residue-toxic effect relationship.

One example of tissue residue-toxic effects linkage is the relationship between failure of Great Lakes lake trout *(Salvelinus namaycush)* to reproduce and bioaccumulation of TCDD and non-ortho substituted PCBs (Mac 1988). Laboratory studies have shown significant mortality of larvae when lake trout ova contain as little as 50 ppt 2,3,7,8-TCDD (Walker et al. 1988). This residue level is found in Lake Ontario lake trout which have not successfully accomplished natural reproduction for many years. On the basis of TCDD toxic equivalents for organochlorine components having the same mode of toxic action, residues in lake trout from Lake Ontario and Lake Michigan may provide a measure of the reduction in sediment contamination necessary to reduce fish tissue concentrations to a presumed reproductive impairment threshold. The same approach can be used for benthic organisms that may have greater inter-site variations in residue levels than do fish because of their more intimate association with sediments.

2.0 DESCRIPTION

2.1 Description of Method

The tissue residue approach involves the establishment of safe sediment concentrations for individual chemicals or classes of chemicals by determining the sediment chemical concentration that will result in acceptable tissue residues. This process involves two steps: 1) linking toxic effects to residues (i.e., dose-response relationships), and 2) linking chemical residues in specific organisms to sediment chemical contamination concentrations (i.e., exposure relationships). Methods to derive unacceptable tissue residues include at least three approaches:

- The water quality criterion-residue approach
- The experimental approach
- The human health approach

Each of these approaches is described briefly below.

Water Quality Criterion-Residue Approach--

A rapid approach for determining acceptable concentrations of tissue residues involves establishing maximum permissible tissue concentrations (MPTC) expected for organisms at the chronic water quality criterion concentration previously established for a specific pollutant. MPTCs, when not available through residue measurements obtained with toxicity tests used as a basis for the water quality criteria, can be obtained by multiplying the water quality criterion by an appropriate bioconcentration factor (BCF) obtained from the literature. When a reliable empirical BCF is not available, the BCF may be predicted from an octanol-water partition coefficient or a bioconcentration kinetic model. Thus, the absence of a water quality criterion for a chemical does not eliminate this approach as long as appropriate chronic toxicity test data are available for the species of interest.

Experimental Approach--

Tissue residue-toxic effects linkages can be established through a series of chronic dose-response experiments or field correlations. Although this approach has the advantage of directly determining the tissue residue-toxic effects linkages, it can be relatively time-consuming and costly to implement for a large number of pollutants. The experimental approach should be used to test the assumptions of the water quality criterion-residue approach and to supplement the existing tissue residue-toxic effects database. The experimental work can be closely coupled with the experiments conducted under the bulk sediment toxicity test approach to deriving sediment quality criteria (see Chapter 2).

Human Health Approach--

Human health risk from consumption of freshwater fish or seafood may be used as the criterion for tissue residue acceptability. A sediment quality criterion for a specific compound can be derived by establishing an acceptable human risk level (e.g., an excess human cancer risk of 1×10^{-5}) and then back-calculating to the sediment concentration that would result in tissue residues associated with this level of risk. The human health approach can generate sediment quality criteria for carcinogenic compounds (e.g., PCBs, dioxins, benzo(a)pyrene) that are lower than those derived from ecological endpoints.

The choice of method to determine a quantitative tissue residue-sediment contamination level relationship depends on the specific pollutants, organisms, and water systems of concern, as well as the regulatory approach (e.g., remedial action, wasteload allocation, Superfund enforcement). The linkage between organism residue and sediment chemical concentration can be made from site-specific measurements of sediment-organism partition coefficients (Kuehl et al. 1987); fugacity or equilibrium partitioning model (Clark et al. 1988); predictions of organism residues; or pharmacokinetic-bioenergetic model predictions of organism residues that result from uptake from food chain, water, and sediment contact (Thomann 1989). The residue approach works best for aquatic ecosystems that are at or close to steady-state with respect to the distribution of chemicals between biotic and abiotic components. Steady-state conditions are common for most sediment contaminants because of their persistence and tendency to exert long-term rather than episodic bioaccumulation and toxic effects.

2.1.1 Objectives and Assumptions--

The objective of this approach is to generate numerical sediment quality criteria based on acceptable levels of chemical contaminants in sediment-exposed biota. This objective is parallel to that of the water quality criteria, except that organism residues provide measures of exposure to chemical contaminants rather than water concentrations of contaminants. By using tissue residues rather than interstitial water concentrations to measure dose, as in the equilibrium partitioning approach (Chapter 5), this method does not require that the organism be at thermodynamic equilibrium with respect to the sediment contamination level. The site-specific residue approach is powerful because it does not require knowledge of bioavailability relationships for each organism in the system. All interaction pathways between sediment and organisms are incorporated in the determination of organism-to-sediment contamination ratios. These can be expressed on the basis of sediment organic carbon-organism lipid for hydrophobic organic chemicals. It is assumed that reduction in sediment contaminant concentrations over time (e.g., as a result of remedial actions, wasteload allocations) will result in parallel reduction in exposure, aquatic organism residues, and, consequently, the potential for toxic effects. It is further assumed that data on residue-to-toxicity relationships can be obtained from laboratory exposures of organisms when such data are not already available and that the route of exposure responsible for residue accumulation does not influence the residue-toxicity relationships.

2.1.2 Level of Effort--

Relatively little effort would be required to generate preliminary sediment quality criteria using MPTCs calculated from existing water quality criteria and BCFs. In the absence of appropriate water quality criteria or

BCFs, the level of effort depends on the availability of non-water quality criteria residue criteria and the complexity of the sediment contaminant mitigation approach to be used. Relatively little effort is required to determine the degree to which sediment contamination concentrations must be reduced for single chemicals in well-mixed systems where fish residues are uniformly unacceptable for human consumption. Much more effort is required for systems having sediment contamination "hot spots" where resident aquatic organisms are eliminated or reduced in number due to a complex mixture of sediment contaminants. Another complexity that could increase the required level of effort is the presence of sediment contaminants that are readily metabolized to chemicals of greater toxicity that are responsible for the observed adverse effects. In some cases, residue-toxic effects data would incorporate the effects of toxic metabolites.

2.1.2.1 Type of Sampling Required--Surface sediment samples must be analyzed for chemical contaminants of interest. Interstitial water composition does not need to be determined because the residues in biota are related to bulk sediment chemical composition. Sediment characteristics such as grain size, organic carbon content, and metal binding capacity are useful for defining sediment-to-biota relationships for different sites within an ecosystem. Biota sampling for residue analysis should include sensitive organisms when toxic effects are a concern, or in the absence of sensitive organisms, organisms whose residues will serve as biomarkers for establishing safe sediment contaminant levels.

2.1.2.2 Methods--The tissue residue approach, as discussed above in Section 2.0, depends on determining residues in aquatic organisms that are unacceptable on the basis of toxicity to the organism or unsuitability for human or animal consumption as food. The linkage of sediment contaminant concentrations to organism residues is possible through a number of approaches including site-specific measurements, equilibrium partitioning-based predictions, and steady-state food chain models. The choice of a specific approach depends on the chemical of concern, the availability of residue-toxic effects data, the contamination history (in-place pollutant problem vs. a continuing or projected sediment contamination problem), and characteristics of the impacted ecosystem. The construction of comprehensive, systematic strategies for all potential sediment contamination assessments will be achieved through further research and development.

Toxicity identification evaluation (TIE) procedures (see Chapter 4) complement the tissue-residue approach. The TIE approach is especially useful if sediment assessment begins without knowledge of the sediment contaminants that are causing toxicity or unacceptable residues in biota. The absence of benthic species or failure of fish eggs to hatch may be attributable to acutely toxic, but non-residue forming, chemicals (e.g., ammonia) in sediments. TIE procedures can distinguish between potential metal, non-polar organic, polar organic, and inorganic chemical sources of toxicity in sediment pore waters or elutriates. These procedures enable a more complete assessment of the significance of residue-associated toxicity in the system.

Once potentially toxic, bioaccumulative contaminants are identified, either in sediment or in aquatic organisms associated through exposure to sediments, the toxicological significance of site-specific sediment-to-biota contaminant partition factors can be assessed. Conservative generic sediment quality criteria can be generated from residue-toxicity relationships by assuming equilibrium partitioning between the binding fractions of organisms and sediments (e.g., lipid and sediment organic carbon for non-polar organic chemicals).

2.1.2.3 Types of Data Required--The tissue residue method requires identification of chemicals in the sediment that are likely to be associated with chronic environmental effects. An indirect method for identifying such chemicals and their locations is to screen aquatic organisms for residues as in the National Dioxin Study or the National Bioaccumulation Study sponsored by U.S. EPA (1987b) Office of Water Regulations and Standards. When toxicity data are not available, either laboratory dose-response experiments or quantitative structure-activity predictions can be used to establish the toxicological significance of the tissue residues. Field surveys that indicate the absence of sensitive organisms in contaminated sediment areas are useful for establishing sediment quality criteria especially if interspecies sensitivities to the chemicals of concern are known. Tissue residues associated with no-effect and the lowest observable effect concentrations are needed when the sediment criterion is not based on a human health standard.

2.1.2.4 Necessary Hardware and Skills--Sediment and tissue analyses require commonly available chemical analytical capabilities. Some chemicals require advanced instrumental analytical techniques, such as high resolution gas chromatography/mass spectrometry.

2.1.3 Adequacy of Documentation--

The use of tissue residues to establish sediment criteria on the basis of human health effects associated with ingestion of contaminated fish has been documented. Methods for using tissue residue-toxicity relationships to establish sediment criteria, although scientifically sound, have not been extensively documented. The various methods for predicting tissue residues in benthos and fish have been well documented.

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

Tissue residue measurements are directly applicable to human risk assessment when the aquatic organism is used as human food. Because of this relationship, the tissue residue method provides a direct link between human health and sediment criteria development. Tissue residues for wildlife and aquatic organisms can be used to assess sediment toxicity when there is an established exposure linkage to the sediment. The tissue residue approach is most advantageous for sediment contaminants that adversely impact organisms such as fish that are not in direct contact with the sediment or its interstitial water. The tissue residue approach is well suited to evaluating sediment quality in systems that have aquatic food chain connections from benthos to birds experiencing eggshell thinning or genotoxic effects. The tissue residue concentration serves as a quantitative measure of sediment contaminant bioavailability, which may differ as a function of ecosystem, sediment, water, food chain, and species characteristics.

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

The tissue residue approach can be used to generate site-specific numerical criteria for non-polar organic chemicals such as PCDDs, PCDFs, and PCBs. Tissue residues of aldrin/dieldrin (U.S. EPA 1980a) and endrin (U.S. EPA 1980b) have been used to establish water quality criteria on the basis of human health risks. The DDT and PCB water quality criteria are based on toxic effects in birds and animals as a function of fish residues (U.S. EPA 1980c,d). Tissue residues of organometallic chemicals such as methyl mercury (U.S. EPA 1984) and elements such as selenium (U.S. EPA 1987a) have been used to establish water quality criteria and/or predict toxic effects. There is some evidence to indicate that metal residues in sediment-dwelling aquatic organisms can reflect both metal bioavailability and potential metal toxicity. Thus, tissue residue-toxicity relationships for some elements could be used as an adjunct to the interstitial water equilibrium partitioning approach.

3.0 USEFULNESS

3.1 Environmental Applicability

3.1.1 Suitability for Different Sediment Types--

There is no limitation to the suitability of this approach for different sediment types, since the method is sensitive to bioavailability differences among sediments. When pelagic organisms are used to assess sediment quality, sediment variability in the water body tends to be averaged.

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

This approach is most applicable to non-polar organics and organometallics that bioaccumulate, are slowly metabolized, and exert chronic toxic effects or present risks to human health. This approach also could work well for chemicals that are metabolized by the organism to nontoxic forms, since the parent compound residue reflects this change in toxic potential. In some cases residues of known metabolites, which are more toxic than the parent compound, can be used to establish residue-toxic effects relationships (Krahn et al. 1986). The approach is not useful for assessing sediment toxicity associated with non-residue forming toxic chemicals such as ammonia, hydrogen sulfide, and polyelectrolytes. Since there is evidence that metal residues in some sediment-dwelling organisms are indicative of both metal bioavailability and potential metal toxicity, sediment quality criteria for metals should be aided by a site-specific tissue residue approach. However, when biological species sequester metals in a nonbiologically available form, tissue residue-toxicity effects linkages may be obscured. The suitability of the method for evaluating additive, synergistic, or antagonistic effects associated with complex mixtures of sediment contaminants depends on the development of chemical mixture toxic dose-response relationships where dose is indicated by tissue residue levels.

3.1.3 Suitability for Predicting Effects on Different Organisms--

The tissue residue approach should not be limited by species unless organism residues cannot be obtained or toxic effects cannot be determined through water quality criteria or bioassays. The key species problem is identification of sensitive species for the sediment contaminants of concern. When adequate comparative toxicity data exist, residues from tolerant organisms may be used to infer sediment criteria for sensitive organisms that are not found in association with the sediment due to toxic effects.

3.1.4 Suitability for In-Place Pollutant Control--

Evaluation of the association of site-specific tissue residues with sediment toxic chemical concentrations provides an established method for in-place pollutant assessment for both human health and ecological risks. Comparison of tissue residues in field-collected organisms to the MPTC would be a direct estimate of ecological risk. The use of resident or caged biota for bioaccumulation potential and toxicity assessments is useful for detection of the most toxic sediments or monitoring of changes in toxicity following remedial action. By weighing the relative toxicity of bioaccumulated pollutants (e.g., by using "dioxin equivalents"), evaluation of tissue residue concentrations can help identify the pollutants most likely responsible for toxicity and their additive contribution to total sediment toxicity. This information could then be used to design the most appropriate and cost-effective mitigation response.

3.1.5 Suitability for Source Control--

The tissue residue approach is well suited for establishing source control. Comparison of the existing or predicted tissue residue levels with MPTCs generates a quantitative estimate of the extent to which a given sediment exceeds or is below a sediment quality criterion. In conjunction with physical transport models, this information can then be used directly to determine acceptable discharge limits, wasteload allocations, or the types of remedial procedures required to achieve acceptable tissue residue levels. The Lake Ontario TCDD-Hyde Park Superfund case example described in Section 1.1 demonstrates the suitability of this approach for establishing source controls. The site-specific nature of this approach provides strong support for establishing controls on existing point and nonpoint sources of sediment contamination.

3.1.6 Suitability for Disposal Applications--

When site-specific sediment-biota contaminant partition coefficients are unavailable, such as for evaluation of proposed disposal operations, the residue approach can be applied by predicting benthic tissue residues from steady-state toxicokinetic bioaccumulation models or by conducting laboratory bioaccumulation tests on the dredged material. If adverse effects on fishes, wildlife, or human health are of concern at such disposal sites, it would then be necessary to apply a trophic transfer or equilibrium partitioning model to predict tissue residues in these higher trophic levels. When the disposal site already has sediments containing the contaminants of concern, residues in existing biota may be used to predict residue levels and toxic effects that would result from additional disposal of similarly contaminated dredged material.

3.2 General Advantages and Limitations

3.2.1 Ease of Use--

The application of sediment quality criteria derived from tissue residues for assessing pelagic or benthic ecological effects is fairly direct. The measured or predicted sediment concentration would simply be compared to the sediment quality criterion derived from MPTCs. The development of a tissue residue toxicity database from laboratory bioassays would allow convenient access to the required biological effects endpoints. Chemical analyses of sediment, total organic carbon, and tissue samples for assessing existing conditions require routine analytical chemistry capabilities that do not present unique problems. One potential difficulty when using tissue residues in field-collected benthos to assess in-place sediments is the difficulty in obtaining sufficient benthic biomass for chemical analysis. This problem can be avoided by conducting laboratory bioaccumulation tests on field- collected sediment or by placing caged benthic organisms in the field.

3.2.2 Relative Cost--

Costs associated with further development of the generic tissue residue approach for sediment quality criteria include 1) development of a residue-toxicity relationship database and 2) validation of the relationships between the MPTC and chronic impacts on aquatic organisms for different chemical classes of sediment contaminants. The cost of applying the method to a particular site, however, depends on the number of sediment and biota samples to be analyzed, the availability of residue-toxicity relationship data, and the difficulty in identifying sensitive organisms. The establishment of a sediment criterion based on fish residue levels acceptable for protection of human health generally incurs low analytical costs when only a few reference sediment sites are needed to characterize the system of concern.

3.2.3 Tendency to be Conservative--

This approach does not tend to be either conservative or liberal for prediction of ecological effects, unless the system responds in a nonlinear manner to reductions in sediment contaminants. In the case of nonlinearity, the tendency would probably be toward conservatism because of the greater bioavailability of more recently introduced sediment contaminants. When human health endpoints are used to generate sediment quality criteria, the criteria may be stricter than necessary to protect resident biota.

3.2.4 Level of Acceptance--

The tissue residue approach is accepted as a basis for regulatory decisions such as the establishment of water quality criteria for the protection of aquatic life and its uses. The direct prediction of chronic toxic effects from measured or predicted tissue residues requires validation before it can be widely endorsed. Since sediment contaminants tend to be long-term exposure problems and can bioaccumulate, residues should be acceptable for sediment criteria development. This approach should be acceptable for identifying sediments associated with a degree of exposure which exceeds that indicated as deleterious in previous experiments.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

The tissue residue approach requires analyses of only sediment and tissue residues when potentially toxic sediment contaminants are known and residue-toxicity relationship data are available. If extensive laboratory work is needed to determine chemical residue-chronic toxicity dose-response relationships for sensitive species, specialized aquatic toxicology capabilities are required. In theory, residue-toxicity based MPTCs can be obtained for all chemicals subject to water quality criteria development.

3.2.6 Level of Effort Required to Generate Results--

The level of effort depends on the number and nature of sediment contaminants, the complexity of the contaminant distribution pattern, and the regulatory application of the method. Some cases will require relatively few analyses of tissue and sediment residues and no toxicity testing to apply the method (e.g., to remedial action decisions, wasteload allocations).

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

Tissue residues that exceed concentrations considered safe for human exposure through seafood consumption require no interpretation when used to set residue-based sediment criteria. However, the degree of interpretation may be very large when evaluating ecotoxicological effects attributed to site-specific measurements of sediment-to-biota chemical partitioning. This interpretation problem exists for all sediment classification methods when applied on a site-specific basis. The presence of unacceptable residues in indicator organisms resident in or linked to an area of sediment contamination can be used without elaborate interpretation to determine compliance of sediments with sediment quality criteria.

3.2.8 Degree of Environmental Applicability--

The use of site-specific tissue residues as quantitative exposure biomarkers eliminates uncertainties associated with chemical bioavailability; exposure duration, frequency, and magnitude; and toxicokinetic/bioenergetic factors. When the tissue residue approach is applied on a generic basis to generate sediment criteria for different chemicals, these uncertainties can be partially addressed through classification of sediments and exposure environments.

3.2.9 Degree of Accuracy and Precision--

Sediment and tissue residue chemical concentrations can be determined accurately and precisely for most chemicals. Most uncertainties in sediment/organism partition coefficients are due to biological variability. Accuracy and precision can be maximized through site-specific investigations of biological factors that influence organism linkage to sediment (through food chain, water, or direct contact) and through refinement of residue-toxicity relationships.

4.0 STATUS

4.1 Extent of Use

Use of tissue residues to establish sediment criteria on the basis of human health effects have been documented. Tissue residues have also been used to derive water quality criteria for the protection of aquatic life and wildlife connected to the aquatic food chain. Tissue residue-toxicity data that may be used for deriving numerical sediment quality criteria for some chemicals already exist in water quality criteria documents, fish consumption advisories, and the peer-reviewed literature. Much aquatic toxicology work in progress or planned for the future could produce the necessary data if residue-based dose measurements are incorporated into research plans.

4.2 Extent to Which Approach Has Been Field-Validated

Sediment TCDD contamination limits have been established for Lake Ontario on the basis of fish tissue residues. This use of tissue residue to generate sediment criteria has been validated through a steady-state model (Endicott et al. 1989) and a laboratory bioaccumulation study (Cook et al. 1989) that demonstrated a linear relationship at steady-state between sediment contaminant concentration and bioaccumulated TCDD in lake trout, regardless of route of uptake. Declines in DDT residues in fish and birds since its use was banned are associated with declining surficial sediment concentrations in the Great Lakes, the Southern California Bight, and elsewhere. Although other examples of studies validating the residue approach for single chemicals are available, its use for complex mixtures of chemicals in sediments to predict sediment safe contaminant concentrations with ecosystem protection in mind has not been validated.

4.3 Reasons for Limited Use

Use of the tissue residue approach has been limited for the following reasons:

- This method is in a developmental stage and has not been formally adopted by U.S. EPA
- Aquatic toxicology has only recently progressed to an understanding of residue-based dose-response relationships for sediment contaminants
- Regulatory agencies, including U.S. EPA, have not yet become committed to systematic establishment and application of sediment criteria methods
- The available and potentially available residue-based toxicity data have not been collated into a database for potential sediment criteria users.

4.4 Outlook for Future Use and Amount of Development Yet Needed

This method can be implemented with a minimal amount of effort in many cases, especially where a single chemical or toxicologically related family of chemicals is of concern. Guidance documents should be written and reviewed. Tissue residue criteria should be accumulated systematically for a database. The use of this method in combination with other sediment classification methods should be considered. Field validation of residue-based ecological effects predictions is essential. All sediment assessment methods should be developed with concern for identification of and application to those chemicals in the aquatic environment that are long-term sediment contaminants having chronic toxicity potential.

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CHAPTER 7. FRESHWATER BENTHIC MACROINVERTEBRATE COMMUNITY STRUCTURE AND FUNCTION

Wayne S. Davis
U.S. Environmental Protection Agency Region V
Environmental Sciences Division
536 S. Clark Street, Chicago, IL 60605
(312 or FTS) 886-6233

Joyce E. Lathrop
Stream Assessments Company
P.O. Box 609
Villa Park, IL 60181

The community structure and function of benthic macroinvertebrates are used extensively to evaluate water quality and characterize impacts in lotic (flowing water) and lentic (standing water) freshwater ecosystems. (Marine benthic community structure is discussed in Chapter 8). Benthic macroinvertebrates are relatively sedentary organisms that inhabit or depend upon the sedimentary environment for their various life functions. Therefore, they are sensitive to both long-term and short-term changes in sediment and water quality. This chapter discusses assessment of benthic macroinvertebrates to determine sediment quality in conjunction with an integrated approach for assessing the quality of the benthic environment. This integrated approach utilizes sediment chemistry, sediment toxicity, and benthic macroinvertebrate community structure and function to evaluate sediment quality, similar to the approaches now used to evaluate surface water quality. The structural assessment relates to the numeric taxonomic distribution of the community, and the functional assessment involves trophic level (feeding group) and morphological assessment. This chapter addresses the specific benthic community assessment methods that are available, or being developed, to complement the chemical and toxicological portions of the sediment quality assessment.

1.0 SPECIFIC APPLICATIONS

1.1 Current Use

Freshwater benthic macroinvertebrate communities are used in the following ways to assess sediment or water quality:

- Identification of the quality of ambient sites through a knowledge of the pollution tolerances and life history requirements of benthic macroinvertebrates
- Comparison of the quality of reference (or least impacted) sites with test (ambient) sites
- Comparison of the quality of ambient sites with historical data to identify temporal trends
- Determination of spatial gradients of contamination for source characterization.

1.1.1 Ecological Uses--

Benthic macroinvertebrate community structure and function assessments have many different applications. Site-specific knowledge of surface water quality, habitat quality, sediment chemistry, and sediment toxicity provide the best context in which to interpret benthic community assessment data. The objectives of each particular study should determine the types of related data necessary. Alone, benthic macroinvertebrates can be used to screen for potential sediment contamination based on spatial gradients in community structure, but they should not be used alone to definitively determine sediment quality. Benthic macroinvertebrates data must be integrated with other available data to determine sediment quality. In a "weight-of-evidence" approach, benthic macroinvertebrates may provide the most important piece of information on sediment quality. Care must be exercised to collect representative samples to minimize natural variations. For example, collections should not be made after floods or other physical disturbances.

Benthic macroinvertebrate community structure and function have been used extensively to characterize freshwater ambient conditions and impacts from various sources. Documented changes in benthic community structure have resulted from crude oil exposure in ponds and streams (Rosenberg and Wiens 1976; Mozley 1978; Mozley and Butler 1978; Cushman 1984; Cushman and Goyert 1984) and heavy metal contamination of lake sediments and streams (Winner et al. 1975, 1980; Wentsel et al. 1977; Moore et al. 1979; Wiederholm 1984a, 1984b; Waterhouse and Farrell 1985). Benthic macroinvertebrates have been used extensively to identify organic enrichment in lentic systems (Cook and Johnson; 1974 Krieger 1984; Rosas et al. 1985) and lotic systems (Richardson 1928; Gaufin and Tarzwell 1952; Hynes 1970; Hilsenhoff 1977, 1982, 1987, 1988). Benthic community responses to pesticides (van Dyk et al. 1975; Webb 1980; Penrose and Lenat 1982; Yasuno et al. 1985), acid- and mine-stressed lotic environments (Simpson 1983; Armitage and Blackburn 1985), thermally stressed water bodies (Crossman et al. 1984), and urban and highway runoff impacts (Smith and Kaster 1983; Dupuis et al. 1985; Denbow and Davis 1986) have also been documented. Chironomidae (midge) larvae were even found to transport substantial amounts of PCBs from contaminated sediments to the terrestrial environment (Larsson 1984).

1.1.2 Regulatory Uses--

Recently, benthic macroinvertebrate communities have been gaining use in U.S. EPA (1988a,b) and state (Ohio EPA 1987a) regulatory programs as the optimal measures of designated use attainment. They are suitable to establish both narrative and numerical instream biological criteria. Among the states implementing or developing instream benthic criteria are Arkansas (Shakelford 1988), Florida (U.S. EPA 1988a), Maine (Courtemanch and Davies 1988), Minnesota (Fandrei, G., 1989, personal communication), Nebraska (Maret 1988), New York (Bode and Novak 1988), North Carolina (Penrose and Overton 1988), Ohio (Ohio EPA 1987a, 1987b), and Vermont (Fiske 1988). Under the Clean Water Act, benthic macroinvertebrates are used for the following:

- Measurement of the restoration and maintenance of biological integrity in surface waters (Section 101)
- Development of water quality criteria based on biological assessment methods when numerical criteria for toxicity are not established [Section 303(c)(2)(B)]
- Production of guidance and criteria based on biological monitoring and assessment methods [Section 304(a)(8)]
- Development of improved measures of the effects of pollutants on biological integrity (Section 105)
- Production of guidelines for evaluating nonpoint sources (NPS) [Section 304(f)]
- Listing of waters that cannot attain designated uses without additional NPS controls (Section 319)

- Listing of waters unable to support balanced aquatic communities [Section 304(1)]
- Assessment of lake trophic states and trends (Section 314)
- Production of biennial reports on the extent to which waters support balanced aquatic communities [Section 305(b)]
- Determination of the effect of dredge and fill disposal on balanced wetland communities (Section 404).

Another important feature of this method is the ability to generate numerical biological criteria for state water quality standards (Ohio EPA 1987a). State development of biocriteria is being encouraged and supported by U.S. EPA, and biocriteria policy and guidance documents are expected to be published by the U.S. EPA Office of Water during FY89-90. In addition to the Clean Water Act regulations, benthic macroinvertebrate community assessments may be applied to Superfund evaluations of onsite and offsite impacts (U.S. EPA 1989a,b). They may also be part of an applicable or relevant and appropriate requirement (ARAR) if a state adopts standards for biocriteria or a "no toxics in toxic amounts" narrative that utilizes benthic macroinvertebrates to determine compliance with those standards.

1.2 Potential Use

The use of benthic macroinvertebrates relating to sediment contamination will be most successful when used with sediment chemistry and toxicity results, as in the "integrated" Sediment Quality Triad approach (see Chapter 9). Benthic macroinvertebrates will best indicate in-place pollutant control needs through a site-specific knowledge of surface water quality, habitat quality, and sediment chemistry and toxicity. Alone, benthic macroinvertebrates can be used to screen for potential sediment contamination and source identification by displaying spatial gradients in community structure, but they should not be used alone to definitively determine sediment quality or develop chemical-specific guidelines. Benthic macroinvertebrate data must be integrated with other available data to determine sediment quality using a "weight-of-evidence" approach.

2.0 DESCRIPTION

2.1 Description of Method

The benthic macroinvertebrate community structure and function assessment involves the following steps:

1. Collection of benthic macroinvertebrates in the field (artificial or natural substrates)

2. Identification to the lowest taxon necessary (varies depending upon the study objectives)

3. Quantification (e.g., taxa richness, number of individuals, indicator organism count, structural indices and ratios, functional characteristics of taxa)

4. Assessment of the relationship with other environmental measurements (e.g., correlations, habitat requirements)

5. Comparison with a "reference" site (e.g., similarity indices, nonparametric analyses)

6. Complete documentation of the study methods, results, and discussion of the relevance of the data.

2.1.1 Objectives and Assumptions--

The primary objective of benthic macroinvertebrate community structure and function analyses is to provide data and information to assist in determining the quality of the sediment/water environment. This determination can then be used for the purposes described above in Section 1.0 (Specific Applications).

It is assumed that benthic macroinvertebrates can provide consistent and accurate assessments of sediment/water quality at a given sample location or water body. Specifically, the following assumptions are implicit in this objective:

- The benthic macroinvertebrates are relatively sedentary, especially compared to fish communities, and they depend upon the sedimentary (or benthic) environment for their life functions

- Chemical and physical perturbations of the sediments or bottom waters affect benthic macroinvertebrates since they are dependent upon the benthic environment for completion of their life cycles, and they are therefore sensitive to changes in sediment and water quality

- Benthic macroinvertebrates physically interact with the sediments to cause chemical exchange between the sediment and the overlying water, and therefore tend to reflect sediment quality as well as water quality

- The optimal use of benthic macroinvertebrates as sediment quality indicators is as part of an integrated sediment quality assessment approach utilizing sediment chemistry, sediment toxicity, and benthic community structure and function.

Equally important assumptions apply to actual benthic macroinvertebrate sampling strategy, collection, identification, data reduction, interpretation of results, and report preparation. It is assumed that all U.S. EPA-supported studies have an adequate quality assurance program plan (QAPP) and that all benthic macroinvertebrate community data are reproducible and collected in a manner to minimize natural variations; the methods must be consistent within each study. Specific QA procedures that should be established early in benthic macroinvertebrate community studies include the following:

- Rationale for sample location selection

- Sample collection methods, sorting, and storage procedures

- Taxonomic proficiency evaluations using either U.S. EPA check-samples from Cincinnati-ERL or State-developed check-samples, in addition to voucher collections from each study area and a list of the taxonomic references used

- Data analysis techniques used to objectively assess the data, including the structural and functional measures

- Nonparametric or parametric (as appropriate) statistical methods used to compare site results.

Each Regional U.S. EPA Quality Assurance Office can provide the details of QAPP requirements.

2.1.2 Level of Effort--

The level of effort to conduct freshwater benthic macroinvertebrate community studies is comparable with chemical/physical water quality measurements and bioassays. However, rapid benthic community assessment techniques can range from 1 to 5 h per site if laboratory identifications are not required. As expected, the greatest time expenditure is in the travel to and from the site and in the sorting and identification of the organisms.

Separating the organisms from debris and sorting the organism can take up to 15 h per sample, with an additional 12 h for identification for very enriched sites with high numbers of individuals among several taxa. More typically, the time spent would be about 3 h for sorting (more time for dredge and artificial substrate samples and less time for dip-net samples), 2 h for preparing the samples (e.g., clearing and then mounting the chironomids on microscope slides), and 6 h for identifying the organisms to the lowest possible taxonomic level. An experienced taxonomist with appropriate keys may average only 2-4 h per site. This typical time equates to about 11 h per site after the samples have been collected. These estimates are only a general guide to the time it may take to perform the identifications, and are meant to help assess potential or actual project costs.

2.1.2.1 Type of Sampling Required--The specific sampling methods to be used are dictated by the study needs. Debate will continue regarding the use of "quantitative" and "qualitative" sampling methods, but each method is acceptable contingent upon how well it will satisfy study objectives, reproducibility of the data, and consistency of collection. Although it is not advisable to critically compare data collected with different sampling techniques, any comparisons made should not involve taxa below the family level. Typically, benthic macroinvertebrate data are quantified by the surface area of the sampler or sediment being collected. However, benthic macroinvertebrates can be quantified in other ways depending upon the objectives of the study. For example, if the objective is to determine the number and types of taxa, rather than the number of individuals within each taxon, then a study using a dip-net in various habitats within the site would be considered quantitative. Examples of programs using data quantified by methods other than surface area of the sampler or substrate include those described by Pollard (1981), Hilsenhoff (1982, 1987, 1988), Cummins and Wilzbach (1985), Bode and Novak (1988), Cummins (1988), Hite (1988), Lenat (1988), Maret (1988), Penrose and Overton (1988), Plafkin et al. (1988), and Shakelford (1988). The success of each sampling effort depends upon a thorough understanding of the data quality objectives of that study and the implementation of a quality assurance program.

In soft freshwater sediments, the most common method used to collect benthic macroinvertebrates is with a grab sampler such as a Ponar (15 x 15 cm or 23 x 23 cm) or Ekman dredge (15 x 15 cm, 23 x 23 cm, or 30 x 30 cm), each of which provides a quantitative sample based upon the surface area of the sampler. The smaller surface area samplers are most commonly used for freshwater studies because of their relative ease of manipulation. The Ekman dredge is not as effective in areas of vegetative debris, but is much lighter than the Ponar and easier to use in softer substrates. Artificial substrates (e.g., Hester-Dendy sampler using several 3-in plates and spacers attached by an eyebolt, or substrate/rock-filled baskets) provide a consistent habitat for the benthic macroinvertebrates to colonize in both soft-bottomed and stony areas. Artificial substrates can be used in almost any water body and have been successfully used to standardize results despite habitat differences (Hester and Dendy 1962; Rosenberg and Resh 1982; APHA et al. 1985; DePauw 1986; Ohio EPA 1987c). The major drawback to artificial substrates is the 4-8 wk period required for instream colonization. At least two visits are required for each study site: one to place the samplers and one to remove them.

A variety of methods for sampling hard-bottomed lotic systems are available. Colonization of substrates and comparisons of the artificial and natural substrate methods have been described (Crossman and Cairns 1974; Beckett and Miller 1982; Shepard 1982; Chadwick and Canton 1983; Peckarsky 1986; Ohio EPA 1987c; Plafkin et al. 1988; Lenat 1988). If quantification by sediment or sampler surface area is needed, a Surber-type square-foot sampler (Surber 1937, 1970) with a 30-mesh (0.589 mm openings) can be used. The traveling kick-net (or dip-net) method, also using a 30-mesh net, can be used to quantify the sample collected by the amount of time spent sampling and the approximate surface area sampled (Pollard and Kinney 1979; Pollard 1981). The Surber-type and kick methods can each be used to provide consistent, reproducible samples but both are limited to wadable streams. The Surber sampler's optimal effectiveness is limited to riffles. Kick or dip-net sampling techniques can be used in all available habitats. Although dip-net samplers have been effectively used to sample riffles and other relatively shallow habitats to determine taxa richness, presence of indicator organisms, relative abundances, similarity between sites, and other information, they do not provide definitive estimates of the number of individuals or biomass per surface area.

For sediment evaluations of lotic systems, a combination of artificial substrate (e.g. Hester-Dendy) and natural substrate (dip-net) sampling is recommended. This combination allows comparison of the benthic macroinvertebrate communities independent of habitat, so that sediment/water quality effects can be better assessed.

2.1.2.2 Methods--Most state environmental regulatory programs have a QAPP describing the field methods and standard operating procedures for collecting and evaluating benthic macroinvertebrates (see previous discussion in Section 2.1.1). This information should be obtained to ensure acceptance and comparability of study results with those obtained by the state agency. If this information is not available, then field methods and standard operating procedures from other existing programs should be used. Since several different collection and analysis methods are used throughout the country depending upon water body type (i.e., lotic vs. lentic), habitat type, substrate type, and familiarity with specific methods, it is not practical to recommend any single sampling method. The only general QA requirement for the use of any one particular method is that the data be reproducible, consistently used within the program, and applicable by different investigators (U.S. EPA 1988a,b).

Sampling Strategy--Sampling strategies have been addressed by Sheldon (1984), Millard and Lettenmaier (1986), and Plafkin et al. (1988). To detect spatial differences in sediment/water quality or to characterize sources of pollution, the best strategy is to collect samples in similar habitats upstream and downstream of suspected pollution sources or other areas of interest for ambient monitoring (e.g., high quality or wild and scenic streams). Preferably two upstream sites and three downstream sites of the suspected pollutant sources should be sampled. However, many programs are limited to only one upstream site and one or two downstream sites. If habitats vary too widely, then artificial substrates should be placed at each site. To complement the artificial substrate data, multi-habitat dip-net sampling should be performed when the substrates are deployed and retrieved.

To best detect temporal trends, a fixed station network should be established near the area of interest and sampled consistently at least one season each year. A reference location should also be sampled at the same times to ensure that differences found in the results can be attributed to changes in water quality near the site. Sampling should be done each year during similar flow conditions and should not be conducted for at least 1-2 wk after a major rainfall.

Seasonal distributions are always a concern for ensuring the collection of a representative sample. Therefore, routine sampling or monitoring is optimal during the seasons indicated in Plafkin et al. (1988), and long-term monitoring should strive for consistent sampling seasons. Participants in the benthic macroinvertebrate discussion group at the 1987 National Workshop on Instream Biological Monitoring and Criteria agreed that the optimal time of year for sampling in lotic systems was during the latter part of the seasons that demonstrate a stable base-flow (normal flow) and temperature regime (Davis and Simon 1988).

Sample Replication--Sample replication is a component of a good QAPP. The following recommendations are somewhat arbitrary, but provide a beginning to implementation of a QA program. When using a new method, at least five replicate samples should be taken per site and analyzed by at least two investigators before the techniques are applied to the program. Coefficients of variation among the samples should be below 50 percent. Although many investigators collect separate replicates and then composite them into one sample, it should be standard practice to analyze each individual replicate for at least 10 percent of the sample sites (or at least one site per study) to check variability. For multihabitat dip-net sampling within a site, the chance of not obtaining a representative sample is greatly decreased, especially since enumeration methods are not likely to be used to quantify the results. If multihabitat samples are routinely composited, then for 10 percent of the sample sites (or at least one per survey), two samples should be collected and analyzed by two investigators. This approach would provide four replicates (two visits by two investigators) from each site sampled.

Statistical derivation of the number of samples required to decrease the variability of the data have been discussed by Weber (U.S. EPA 1973), Green (1978), Merritt et al. (1984), and Resh and Price (1984). These methods rely on prior knowledge of the variability of the data. This prior knowledge is often not available or practical to obtain from a programmatic view (e.g., the cost of initial sampling to estimate variability and required number of replicates may be prohibitive). Another problem with statistically determining the number of samples needed is the assumption that the data follow a specific distribution such as normal or log-normal, which is not necessarily true for biological samples. Also, the variability, as measured by the variance or standard deviation, would be different for each descriptive index analyzed (e.g., number of taxa vs. number of individuals).

Field Methods--Field sampling methods are adequately addressed in many manuals including the U.S. EPA (1973) biological field methods manual, the ASTM (1988) methods for sampling benthic macroinvertebrates, Ohio EPA's (1987c) field methods manual, Standard Methods (APHA et al. 1985), U.S. EPA's rapid bioassessment protocols (RBP) (Plafkin et al. 1988), and U.S. EPA's (1987) Superfund field compendium. The following decisions will need to be made once the sample gear is chosen:

- **Whether or not samples will be picked from debris and sorted in the field**
- **Which preservative should be used**
- **Whether or not a stain (e.g., rose bengal) will be added to the sample to facilitate separating the organisms from debris**

- Whether or not the samples need to be shipped and whether or not they require a chain-of-custody form (as in Superfund samples)

- The type of sample containers.

Sieving of samples is addressed in the next paragraph. If sieving is performed in the field, a #30-mesh sieve (0.595-mm openings) is recommended, with the materials that pass through then sieved through a #40-mesh sieve (0.425-mm openings). Separation of organisms and debris should be performed in the laboratory. Depending on the desired level of effort for the study, however, separation can be done in the field by placing the sample in a white enamel pan to provide a bright background to see the organisms. Rose bengal (200 mg/L) can be used to stain the organisms to facilitate separation from detritus and other debris (APHA et al. 1985). For routine benthic sampling, special fixatives are not necessary and the organisms should be preserved in a 70 percent ethanol solution. Formalin, which is an excellent fixative, is no longer recommended because of health effects. The organisms should always be collected in plastic, shatter-proof containers.

Sorting--There are many discussions elsewhere of techniques for sample sorting and preparation of slides for identification. For example, Weber (U.S. EPA 1973), Pennack (1978), Merritt et al. (1984), and APHA et al. (1985) offer excellent guidance for sample sorting. Hynes (1970, 1971) states that the earlier stages of benthic organisms are retained by a 0.2-mm mesh size (approximately the size of a #75 standard sieve), and APHA et al. (1985) and Weber (U.S. EPA 1973) define benthic organisms by a mesh size of 0.595 (standard sieve #30), which is now standard practice. However, some types of chironomids and other small benthos pass through the #30-mesh sieve but are retained by the #40-mesh sieve. It is therefore recommended that samples be passed through a #30-mesh sieve and that the materials washed through be passed through a #40-mesh sieve; the material retained in both sieves should then be sorted (Ohio EPA 1987c). Once the material is washed through the sieves, the organisms should be separated from the vegetation and other debris in a white enamel pan. As the materials are separated, the organisms can be placed in different vials for the major taxa.

Midge Preparation--Slide preparation for the genus and species identification of the chironomids can involve either wet mounts or permanent mounts. In either case, the head capsule is the primary part of the organism used in their taxonomy, and must be cleared of pigmentation for best identification. Simpson and Bode (1980) and Ohio EPA (1987c) recommend clearing the organisms in either a cold solution of 10 percent potassium hydroxide (KOH) overnight, or a heated (not boiling) solution of the same for no more than a few hours. Once the samples are rinsed in water (less than 0.5 h), the organisms can be examined on a wet or permanent mount. The wet mount uses only a few drops of water and a cover slip. The midges need to be dehydrated in successive solutions of 70 percent and 95 percent ethanol if a permanent mount is to be made. The prepared midges can then be permanently mounted using a number of media, including Euparol before the cover slip is applied. Great care should be taken to ensure the ventral side of the head capsule is fully visible before making the mount permanent. Generally, only the voucher specimens need to be permanent mounts, while the other midges can be identified by using wet mounts. Weber (U.S. EPA 1973), Pennack (1978), Simpson and Bode (1980), Merrit and Cummins (1984), and Ohio EPA (1987c) provide detailed guidance on this subject.

Taxonomy--The level to which the taxonomic identifications should be taken is dependent upon the objectives of the study. For a reconnaissance or screening survey, it is generally not necessary to go beyond family (Illinois EPA 1987; Hilsenhoff 1988; Plafkin et al. 1988; Resh 1988). For studies attempting to identify designated use impairment or evaluate impacts from a specific source, the minimum level of taxonomic detail should follow recommendations of Ohio EPA (1987c). Ohio EPA has successfully implemented numeric biocriteria based on this taxonomic detailing. The focus is on differentiating taxa that are better water quality indicators and for which taxonomic keys and expertise are readily available. The level of taxonomic detailing must be consistent within the program and applied for each sample site. Species-level identifications for all organisms are not necessary for a successful program and they commonly depend upon the availability of local keys. National keys available for genus-level identifications include Merritt and Cummins (1984) for insects, Pennack (1978) for all common invertebrates, and Klemm (1985) for annelids (oligochaetes and leeches). Regional U.S. EPA or state biologists should be contacted to determine which of the hundreds of other taxonomic keys are available for specific taxa, both nationally and regionally.

2.1.2.3 Types of Data Required--The types of data analyses that are required to meet program objectives directly affect the types of data required. A list of the families of taxa present may be sufficient to meet some program objectives. Under other circumstances, species-level taxonomy and enumerations may be required. The necessary data to conduct different types of analyses can be obtained from the following discussion of data analysis methods.

One of the most inconsistent and perplexing aspects of a freshwater benthic macroinvertebrate community assessment is the numeric representation and analysis of the data collected. Structural community measures such as richness values, diversity and biotic indices, and enumerations have been used almost exclusively. Indicator organisms have been used to establish many of the biotic indices but also have the potential to differentiate among types of impacts. Recently, functional community measures based on feeding groups such as shredder, collector, scraper, and predator (Cummins and Merritt 1984) have gained wider application and acceptance due to their sensitivity in detecting system perturbation on food resources. Sediment and water quality assessments based on the benthic macroinvertebrate community should use a complementary mix of both structural and functional measures. Discussions of various data analysis techniques can be found in Hawkes (1979), Cairns (1981), Washington (1984), and Resh (1988).

Diversity Indices--When diversity indices were introduced, they were used widely because of their ability to reduce the complex benthic community measurements into a single value that could be used by nonbiologist decision-makers. Diversity indices are based on measuring the distribution of the number of individuals among the different taxa, and use methods that result in enumerations by surface area. The most common diversity index used for water quality studies is the Shannon, or Shannon-Wiener index (Shannon and Weaver 1949) as shown below:

$$\text{Shannon's } H' = \sum_{i=1}^{s} (n_i/n) \ln (n_i/n)$$

where:

n_i = Total number of individuals in the i^{th} taxon
n = Total number of individuals
s = Total number of taxa.

[Washington (1984) provides a good explanation of how the index derived the name Shannon-Wiener rather than Shannon-Weaver index]. Theoretically, higher community diversity indicates better water quality (Wilhm 1970). However, low diversity may be caused by factors other than water quality impacts, such as extremes in weather (floods or droughts), poor habitat, or seasonal fluctuations. Although diversity indices such as the Shannon-Wiener index still remain in widespread use (Washington 1984), their limitations in accurately addressing a variety of perturbations has decreased their reliability (Cooke 1976; Hilsenhoff 1977; Hughs 1978; Chadwick and Canton 1984; Washington 1984; Mason et al. 1985; Resh 1988). Kaesler et al. (1978)

demonstrated that the popular Shannon's Index was actually not the preferred index for aquatic ecology studies and recommended the use of Brillouin's (1962) Index. Resh (1988) reported that diversity indices showed varied results in detecting changes in water quality and that they are not the optimal measures of water quality. However, diversity indices can provide additional information as to the community composition and should be reported if the data are available. Reliance upon these indices as the only, or predominant measure upon which water pollution control decisions are based is not valid. Washington (1984) provides an outstanding review of the history and uses of diversity indices.

Biotic Indices--Biotic indices utilize pollution tolerance scores for each taxon, weighted by the number of individuals assigned to each tolerance value. If desired, relative abundance measures can be used in biotic indices. An example of a widely used biotic index (Hilsenhoff 1977, 1982) is as follows:

$$\text{Biotic Index} = \sum_{i=1}^{s} n_i a_i / n$$

where:

n_i = Number of individuals in taxon i
a_i = Tolerance value assigned to taxon i
n = Total number of individuals in the sample.

Tolerance values can be found in Hilsenhoff (1987) or can be generated by regional-specific knowledge of the organisms' tolerances. Typical ranges of organism index values are 0-5, 0-10, or 0-11, with the higher numbers indicating greater tolerance to pollutants. Community indices are generally limited to lotic systems impacted by organic enrichment (Woodiwiss 1964; Chandler 1970; Hilsenhoff 1977; Murphy 1978; DePauw et al. 1986) or other general perturbations (Hawkes 1979). Biotic indices based on a specific population, rather than community, are addressed in the "Indicator Organism" discussion below. Although the first widely applied biotic index in this country was developed by Beck (1955) for Florida streams, the Hilsenhoff Biotic Index (Hilsenhoff 1977, 1982) has gained great popularity and has been updated to revise the scoring system from a range of 0-5 to 0-11 (Hilsenhoff 1987) and to include a family-level biotic index (Hilsenhoff 1988). Because the biotic indices rely heavily on known pollution tolerances of the taxa, Washington (1984), Mason et al. (1985), and Hawkes (1979) preferred the biotic indices over the diversity indices for water quality assessments. The success of the Hilsenhoff Biotic Index prompted its use, or modifications of it, in several state programs (e.g. Wisconsin, Illinois, New York, North Carolina). Unfortunately, tolerance values are not available for many taxa because they tend not to exhibit water quality preferences, and the assessments are generally limited to organic enrichment. Washington (1984) provides an outstanding review of the history and uses of these indices.

Indicator Organisms--Indicator organisms have played a key role in the development of biotic indices for both lotic and lentic systems. One of the first classifications based on indicator organisms was done in the Illinois River by Richardson (1928). Simpson and Bode (1980), Bode and Simpson (unpublished), and Rae (1989), among many others, utilized Chironomidae as indicator organisms for a variety of toxicants in stream systems. Hawkes (1979) provides an excellent review of the use of benthic macroinvertebrates for stream quality assessments, and Wiederholm (1980) does the same for lake systems. Data analyses for benthic macroinvertebrates in lentic systems have not been as progressive as those in lotic systems with regard to composite indices, and have relied extensively on enumerations, diversity indices, richness values, and indicator organisms (Fitchko 1986). Howmiller and Scott (1977), Krieger (1984), and Lauritsen et al. (1985) used oligochaete communities to establish a Great Lakes trophic index. Lafont (1984) also used oligochaetes to indicate fine sediment pollution. Brinkhurst et al. (1968) and Winnell and White (1985) used chironomids to develop a similar index for the Great Lakes, and Courtemanch (1987) classified Maine lakes using chironomid larvae similar to the studies of Saether (1979) and Aagaard (1986) in European lakes. Hart and Fuller (1974) presented pollution ecology data for a number of freshwater benthic macroinvertebrates as did U.S. EPA's pollution tolerance information series on Chironomidae (Beck 1977), Trichoptera (caddisflies) (Harris and Lawrence 1978), Ephemeroptera (mayflies) (Hubbard and Peters 1978), and Plecoptera (stoneflies) (Surdick and Gaufin 1978). Washington (1984) also reviewed population-based biotic indices.

Richness Measures--Richness measures are based on the presence or absence of selected taxa. Commonly used measures include the total number of taxa, number of EPT (Ephemeroptera, Plecoptera, and Trichoptera), and the number of families. The higher the richness value is, the better the quality of the system. Richness measures have been shown to have low variability and high accuracy in identifying impact (Resh 1988) and should be applied in each study.

Enumerations--Enumerations involve obtaining a sample quantified by surface area to obtain specific abundances of each taxon. Examples include the number of total individuals, number of EPT individuals, ratios of number of individuals within a taxon to the total number of individuals (Ohio EPA 1987a; Resh 1988), and ratios of the number of individuals within one taxonomic group (e.g., EPT) individuals to number of individuals within another taxonomic group (e.g., Chironomidae) (Plafkin et al. 1988; Resh 1988). Interpretation of the enumeration ratios can be difficult without prior validation. Most possible enumerations comparing individual taxa to the total number of individuals are done for many studies, although the results may not be presented. The percent contribution of the individuals within a taxon at a sample site can be compared with the percent contribution at the reference sites to detect a change in community structure. Resh

(1988) concluded that the seven common enumerations he tested had extremely high variability and unacceptably low accuracy in detecting various impacts, and suggested that they are not as useful for detecting environmental change as richness measures or the family biotic index. Although the measures Resh (1988) used may not be optimal for widespread use, they may still provide insight into changes in the community structure. Ohio EPA (1987a) has successfully used enumerations for the percentage of mayflies, caddisflies, Tanytarsini midges, tolerant organisms, and "other" dipterans combined with non-insect individuals as a basis for their state biocriteria.

Similarity Indices--Community similarity indices measure the similarity between benthic communities at a reference and a study site, with high similarity indicating little change, or impact, between the two sites. The use of similarity indices has been reviewed by Brock (1977) and Washington (1984). The simplest indices to apply are those that use only the types of taxa found, not the abundance of the organisms within each taxon. The Jaccard Index (1908) and Van Horn's Index (1950) are examples of the simpler indices. Van Horn's Index used by Ohio EPA (1987c) is as follows:

$$\text{Similarity (c)} = 2w/(a+b)$$

where:

a = Number of taxa collected at one site
b = Number of taxa collected at the other site
w = Number of taxa common to both stations.

A value over 6.5 or 7.0 indicates good similarity. Plafkin et al. (1988) utilize the Jaccard Index in the rapid bioassessment protocols (RBPs). Other indices such as the percent similarity (Brock 1977) and the Bray-Curtis (1957) utilize the abundance of organisms.

Functional Information--Community function measurements based upon habitat, trophic structure, and other ecological measures were described by Kaesler et al. (1978) and used by Rooke and Mackie (1982a) as the "ecological community analysis" (ECA). Rooke and Mackie (1982b) reported the ECA to provide more information on environmental quality than diversity or biotic indices, but the ECA was very time-consuming and not practical for rapid assessments. However, Cummins and Wilzbach (1985) and Cummins (1988) describe a rapid assessment method based on sampling coarse particulate organic matter and determining the functional feeding groups described in Merritt and Cummins (1984). This method is being used in the RBPs (Plafkin et al. 1988). Rabeni et al. (1985) also described the usefulness of a functional feeding group approach to provide a "more ecologically sound classification of water quality" during their development of a biotic index for paper mill impacts. Another useful measure of function is observations of the incidence of morphological deformities in benthic macroinvertebrates, similar to the observations made for the Karr's index of biotic integrity (IBI) for fish (Karr et al. 1986). Deformities have been associated with exposure of metals and organic compounds to Chironomidae (Cushman 1984; Cushman and Goyert 1984; Wiederholm 1984b; Warwick 1985; Warwick et al. 1987) and Trichoptera (Simpson 1980; Petersen and Petersen 1983).

Composite Indices--Composite indices combine selected structural or functional measures, or "metrics," in a cumulative scoring system, as was done with the IBI for the fish community (Karr et al. 1986). Ohio EPA (1987c) developed a similar index for invertebrates using the following 10 structural metrics, adjusted for drainage area size, to derive a final Invertebrate Community Index (ICI) score:

1. Total number of taxa

2. Total number of mayfly taxa

3. Total number of caddisfly taxa

4. Total number of dipteran taxa

5. Percent mayflies

6. Percent caddisflies

7. Percent Tribe Tanytarsini midges

8. Percent other dipterans and non-insects

9. Percent tolerant organisms

10. Total number of qualitative EPT taxa

The ICI score is directly related to Ohio EPA's numeric biocriteria for designated use attainment, and was developed using artificial and natural substrate data for 232 "least-impacted" reference sites. A statistical validation of the ICI using a factor analysis technique showed high correlations between the factor analysis scores and the ICI scores and little redundancy between the metrics (Davis and Lubin 1989).

U.S. EPA (Plafkin et al. 1988) developed a composite index for rapid assessments in lotic systems using the following two functional and six structural metrics:

1. Taxa richness

2. Modified Hilsenhoff biotic index

3. Ratio of scrapers and filtering collectors (functional)

4. Ratio of EPT and Chironomidae abundances

5. Percent contribution of dominant taxon

6. EPT index

7. Community similarity index

8. Ratio of shredders to total number of organisms (functional).

These RBPs are developed by conducting single-habitat (riffle) dip-net sampling. The scores are based on a percentage of the metric values found at a reference site, rather than comparison of the results based on "optimal" values for each metric. U.S. EPA Region V is currently developing such "optimal" metric values. The RBPs are flexible and can be modified for different geographical locations, as evidenced by the use of different metrics in Arkansas (Shakelford 1988) and New York (Bode and Novak 1988). The success of the RBPs is in the use of the composite index for rapid assessments that allows for three levels of taxonomic work (i.e., order, family, or genus/species levels). Order and family taxonomy do not require laboratory taxonomy and may be done in the field. The RBPs normally use single habitat (riffle) sampling and are limited to a 100 organism count in the field. However, they can be adapted for most program uses, for example by employing multihabitat sampling and/or various count limitations. To be applicable to a state's program, the RBPs should undergo a rigorous validation effort within that state.

Statistical Approaches--Various statistical approaches have been applied to determine whether the benthic community at a study site varies from that at a reference or other site. Depending upon the chosen endpoints of the study, rigorous statistical analysis may not be necessary. For instance, if the endpoint is the number of taxa or richness measures, the variability is generally quite low and accuracy quite high. In this case, the differences between two communities would need to be evaluated based on study objectives. A "statistical" difference between two communities will not indicate whether more subtle changes in community composition are occurring or whether mitigation may be warranted before a statistical change occurs. Sometimes when that change occurs, it is too late to protect the community. The same data evaluation procedures apply to both the marine and freshwater systems. The reader is referred to the statistical discussion in Chapter 8 (marine benthic community structure).

Bivariate and multivariate analysis are often applied in benthic studies to define relationships between and among variables. Examples of these analyses include analysis of variance (ANOVA), correlations, regressions (including multiple regressions), and the two sample t-test. A major drawback to these methods is the assumption that the data follow a statistical distribution such as a normal or log-normal distribution. This assumption is often invalid when dealing with biological populations and communities.

Alternatively, nonparametric analyses may be conducted. Such analyses are not based on assumptions about a specific distribution of the data. Examples of such tests include the chi-square test, binomial tests, rank correlations, or tests comparable to the t-test such as the Mann-Whitney test. Whichever statistical methods are employed, all data assumptions must be clearly stated and objectives known.

2.1.2.4 Necessary Hardware and Skills--The hardware needed for field collection includes samplers (e.g., dredges, dip-nets), sieves, benthic macroinvertebrate containers, forceps, white enamel pans, ethanol preservative, and appropriate personal gear (e.g., hip boots or chest-waders, life-vest if needed, and first aid kits). For the laboratory, standard biological laboratory equipment should be available, such as microscopes (both dissecting and compound), forceps, microscope slides and cover slips, ethanol, potassium hydroxide, mounting media, and sieves. A personal computer (containing a 20 MB or larger hard drive) is important for storing and analyzing the data.

Trained benthic macroinvertebrate field biologists and taxonomists are needed for benthic community assessments. At least one should be proficient at identifications beyond the family-level. That taxonomist should remain involved until the proficiency of the identifier in reaching family-level identifications is assured. A minimum of a Master of Science degree in a related discipline is usually required for the taxonomist to have learned the necessary skills. However, adequate training is commonly available through taxonomy courses and workshops that can provide the necessary proficiency without an advanced degree. A demonstration of proficiency by accurately identifying a check sample prepared by U.S. EPA or a state agency is important. A trained benthic ecologist is necessary to compile and interpret the data. Although it would be ideal if the benthic ecologist had a rigorous statistical background, consultation with a statistician should be adequate.

2.1.3 Adequacy of Documentation--

There is ample documentation of both field methods and analytical techniques. The *Journal of the North American Benthological Society* is a prime source of this information, as is technical exchange at professional meetings. Furthermore, there is a large volume of published and unpublished material that documents use of this method.

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

This method is directly applicable to the protection of aquatic life since it is based on direct measurements of benthic macroinvertebrates. This method is directly applicable to the protection of those aquatic organisms (e.g., fish) and wildlife that directly feed on benthic macroinvertebrates (e.g., small mammals and wading shorebirds). It is indirectly applicable to other wildlife that depend upon benthos at other levels in the food chain. This method is also indirectly applicable to the protection of human health, since benthic macroinvertebrates can serve as indicators of toxicant impacts that may affect humans via bioaccumulation pathways.

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

This method is used in conjunction with sediment toxicity and chemistry data to characterize toxicant impacts and assist with determining the appropriate levels at which the toxicants should be controlled. However, by itself, this method would not be used to generate chemical-specific criteria.

3.0 USEFULNESS

3.1 Environmental Applicability

Benthic macroinvertebrates have been routinely used to assess environmental quality in a variety of geographical areas and ecoregions as was discussed in Section 1.0.

3.1.1 Suitability for Different Sediment Types--

Assessment of the freshwater benthic macroinvertebrate community structure is well suited for evaluating different sediment types, since the benthos inhabit all substrates (Merrit and Cummins 1984). Comparisons should be made among benthic communities of similar substrate since different types and numbers of organisms will inhabit different types of substrates.

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

Benthic macroinvertebrate communities are routinely used to assess potential impacts caused by many different chemicals or classes of chemicals. In addition to the uses described in Section 1.1.1 of this chapter, many benthic organisms are used to indicate stresses from specific chemicals or classes of chemicals (Brinkhurst et al. 1968; Hart and Fuller 1974; Saether 1979; Simpson and Bode 1980; Wiederholm 1980; Bode and Simpson unpublished; Winnell and White 1985; Aagaard 1986; and Fitchko 1986).

3.1.3 Suitability for Predicting Effects on Different Organisms--

The use of benthic macroinvertebrates as indicator organisms has already been discussed. Benthic macroinvertebrates can be used to predict the effects upon other aquatic organisms because if the benthic macroinvertebrate community is impacted, then the impact is likely to be, or already has been, detrimental to other organisms.

3.1.4 Suitability for In-Place Pollutant Control--

Benthic macroinvertebrates will best indicate in-place pollutant control needs through a site-specific knowledge of surface water quality, habitat quality, and sediment chemistry and toxicity. Alone, the benthic macroinvertebrates can be used to screen for potential sources of sediment contamination based on spatial gradients in community structure, but they should not be used alone to definitively determine sediment quality or develop chemical-specific guidelines. The benthic data must be integrated with other available data to determine sediment quality using a "weight-of-evidence" approach.

3.1.5 Suitability for Source Control--

Benthic macroinvertebrates have been extensively used for source characterization and control in many of the state and U.S. EPA monitoring programs involving spatial surveys upstream and downstream of suspected sources (Ohio EPA 1987a; Bode and Novak 1988; Courtemanch and Davies 1988; Fiske 1988; Maret 1988; Penrose and Overton 1988; Shakelford 1988; U.S. EPA 1988a,b; Fandrei 1989). If a detrimental change is detected in the benthic macroinvertebrate community and that change can be attributable to a source, then control measures can be implemented through the NPDES permit program. Many states aggressively pursue this action.

3.1.6 Suitability for Disposal Applications--

The discussion presented in Section 3.1.6 of Chapter 8 (marine benthic macroinvertebrate community structure) is applicable to fresh water. Recently benthic community assessments have been required by U.S. EPA (1989c) Region V, as stated in the *Draft Interim Guidance for the Design and Execution of Sediment Sampling Efforts Relating to Navigational Maintenance Dredging in Region V - May 1989*. In this guidance, benthic macroinvertebrate assessments are advised for areas that are suitable for open-lake disposal or for sediments that are difficult to characterize. All benthic community assessments will be made in concert with sediment chemistry and toxicity evaluations.

3.2 General Advantages and Limitations

The advantage of using the benthic macroinvertebrates community assessment approach to determining sediment quality is that it provides an economical and accurate indication of the health of the system under study, and it is based on direct observation rather than theoretically derived data. The major limitation is the difficulty in relating the findings to the presence of individual chemicals and specific concentrations of those chemicals for numeric in-place pollutant management. This method should be integrated with sediment chemistry and toxicity information.

3.2.1 Ease of Use--

The equipment requirements for benthic surveys is minimal and inexpensive compared to those for chemical/physical analyses or even toxicity tests. The organisms are easy to obtain, but difficult to sort and identity. All materials needed for benthic assessments are easily obtained through chemical and biological supply companies and require no special mechanical setup or calibration.

3.2.2 Relative Cost--

The cost for benthic macroinvertebrate assessments is economical compared to that for chemistry or toxicological evaluations. Ohio EPA (1987a) provided a cost of about $700 to conduct a benthic assessment at one sample site. However, this cost included overhead (e.g., rent, office equipment), all travel expenses, time spent in the field, and report preparation. Ohio EPA conducts artificial substrate (composite of five substrates) sampling in addition to natural substrate (multi-habitat) sampling at each site. Their cost of $700 was quite economical compared to chemical/physical testing ($1,500) or bioassay testing ($3,000 to $12,000) for each site.

The most expensive items are the samplers and the microscopes to identify the organisms. However, most state programs and contractors have this equipment available for other program needs. The fieldwork can be conducted during the time it takes to collect a sediment sample. The most time-consuming aspect is the laboratory sorting and identifications, which may average 11 h per site. However, this process compares favorably with the amount of time required to set up and run a toxicity test or to prepare and analyze chemical variables.

3.2.3 Tendency to be Conservative--

The benthic macroinvertebrate community assessment provides a conservative measure, since the community is responding to both temporal and spatial perturbations. There are few chances, if any, of obtaining a result indicating a high quality community when an impact occurs. Because of influences other than sediment/water quality, it is more common to observe an impacted community when there is no sediment/water quality impact. Although the primary focus is on community level information, changes in individual populations could also be addressed. However, the ecological significance of population changes may not be evident until the community is affected.

In a review of surface water chemistry and benthic macroinvertebrate community assessments from 431 sites in Ohio, benthic macroinvertebrates were more sensitive (conservative) indicators of water quality (Ohio EPA, personal communication). In 35.6 percent of the sites, the benthic assessment revealed impacts not detected by chemical analyses. In 58.1 percent of the sites, the chemical and biological assessment supported one another. Only 6.3 percent of the sites did not have a benthic impact when the chemistry indicated that there would be one.

3.2.4 Level of Acceptance--

Benthic macroinvertebrate community assessments of sediment/water quality have been used in freshwater systems since the early 1900s

(Richardson 1928). Most of the methods employed today have been widely accepted for use, although the use of function measurements is not as well documented. Perhaps the single most important demonstration of the level of acceptance of benthic assessments is the growing regulatory use and establishment of numerical biological criteria in state water quality standards.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

The only special pieces of equipment required are the samplers and sieves, which are easily obtained from biological supply warehouses. Most biological laboratories will have dissecting and compound microscopes, chemical reagents, microscope slides and cover slips, forceps, and any other materials needed. The laboratory's capability to identify benthic macroinvertebrates is less common. Taxonomy is not a widespread skill and is more likely to be found in consulting firms than in analytical laboratories.

3.2.6 Level of Effort Required to Generate Results--

Depending upon the study objectives and level of effort needed, results can be generated in written form in as quickly as 1 day (Plafkin et al. 1988) or in several months. For example, Ohio EPA processes over 500 individual benthic samples each year, identifies the organisms, and prepares reports for regulatory use in less than 1 year, with fewer than three full-time employees in their benthic macroinvertebrate unit. The critical period is the turnaround time for the taxonomy. With artificial substrates, an additional 6-wk colonization period is required; unless a rapid assessment or moderate sized study is done, a written report including interpretation of results will require between 6 mo and 1 yr.

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

It is never advisable to have an individual without training in benthic ecology interpret benthic data. Once the benthic ecologist provides a report with recommendations, the results can be easily implemented into a management strategy. Although several numerical indices that appear simple to use are available, data interpretation relies upon all of the information generated for a study, including chemical, physical, and toxicological measurements, as well as indicator organisms and function measures.

3.2.8 Degree of Environmental Applicability--

Benthic macroinvertebrate community structure and function is used extensively to evaluate sediment and water quality and characterize impacts in lotic and lentic freshwater ecosystems.

3.2.9 Degree of Accuracy and Precision--

Since benthic macroinvertebrates are measured directly, this method is highly accurate for characterizing sediment/water quality effects upon aquatic life. There is little chance, if any, that a high quality community will be indicated when an impact actually occurs (Type II error with a null hypothesis of no community change). Because of influences other than sediment/water quality, it is more common to indicate an impacted community when there is no sediment/quality impact (Type I error with a null hypothesis of no community change). For environmental pollution control, a Type II error is much more serious than a Type I error, which is conservative. To reduce the possibility of a Type II error, the interpretation of the data (including chemistry and toxicity) must be done by a trained benthic ecologist. Resh (1988) reviewed the levels of accuracy and precision for several of the data analysis techniques.

To ensure as much accuracy and precision in the data as possible, a detailed quality assurance program plan should be established and followed. Careful and consistent field and laboratory protocols are necessary. It is also necessary to sample during optimal conditions, which can minimize the effects of natural variations in the data. However, the natural variability, especially seasonal, is reduced when using a community-level approach rather than a population-level approach.

4.0 STATUS

Sections 1.1 (Current Uses) and 3.0 (Usefulness) describe the status of the discipline.

4.1 Extent of Use

This method is widely used in both regulatory and nonregulatory sediment and water quality programs. It has been used to assess impacts due to organic enrichment and a variety of chemical classes in both lotic and lentic systems. Benthic macroinvertebrate community assessments are the most widely used instream biological measures in state water quality programs.

4.2 Extent to Which Approach Has Been Field-Validated

Since it is an *in situ* study, field validation occurs when the approach can consistently and accurately assess environmental quality. Most benthic studies employ reference stations and rely upon other environmental data to validate the method. The documentation provided in this paper should present adequate documentation of the method's validity.

4.3 Reasons for Limited Use

Benthic macroinvertebrate community assessments are very common in freshwater systems because of their relatively low cost and high information output.

4.4 Outlook for Future Use and Amount of Development Yet Needed

The outlook for the future use of benthic macroinvertebrate community structure and function in sediment quality assessment is very good because of the recognition that benthic macroinvertebrates provide substantial information that the chemistry and toxicity data alone cannot provide. With the Clean Water Act mandate to maintain and restore biological integrity, benthic community assessments can help determine whether sediment quality is impairing the designated uses and biotic integrity. With the increasing reliance upon numerical biocriteria, additional sediment quality problems will be identified. The area where development is most needed is in combining benthic community assessments with chemical and toxicological data in an integrated approach for assessing sediment quality. In addition, the functional measures, which also hold much promise for sediment assessments, need to be validated more thoroughly.

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CHAPTER 8. MARINE BENTHIC COMMUNITY STRUCTURE ASSESSMENT

Betsy Day, Gary Braun, and Gordon Bilyard
Tetra Tech, Inc.
11820 Northup Way, Suite 100E
Bellevue, WA 98005
(206) 822-9596

Benthic communities are communities of organisms that live in or on the sediment. In most benthic community structure assessments, primary emphasis is placed on determining the species that are present and the distribution of individuals among those species. These community attributes are emphasized largely for pragmatic reasons. Whereas it is relatively simple to collect, identify, and enumerate benthic organisms, it is very difficult to determine first hand the spatial distributions of species and individuals within the benthic habitat, or the functional interactions that occur among the resident organisms or between the resident organisms and the abiotic habitat. Hence, information on benthic community composition and abundance is typically used in conjunction with information in the scientific literature to infer the distributions of species and individuals in three dimensional space and the functional attributes of the community. Because all of the major structural and functional attributes of benthic communities are affected by sediment quality in generally predictable ways, benthic community structure assessment is a valuable tool for evaluating sediment quality and its effects on a major biological component of marine, estuarine, and freshwater ecosystems.

Benthic habitats may be broadly divided into hard-bottom habitats and soft-bottom habitats. Many types of each exist in marine, estuarine, and freshwater ecosystems. Hard-bottom habitats include rocky shorelines and bottoms of lentic and lotic systems, rocky intertidal and subtidal habitats in marine and estuarine systems, and coral reefs. Soft-bottom habitats include mud and sand habitats in marine, estuarine, and freshwater systems; marine, estuarine, and freshwater macrophyte beds; freshwater wetlands, and estuarine salt marshes. Each of these habitats requires different sample collection methods and different survey design considerations. The emphasis of this chapter is on assessments of marine benthic community structure in soft- bottom habitats as an indicator of sediment quality. Freshwater benthic invertebrate community structure is discussed in Chapter 7.

1.0 SPECIFIC APPLICATIONS

Assessment of benthic community structure is an *in situ* method that can be used alone, as part of other approaches [e.g., Sediment Quality Triad (see Chapter 9) and Apparent Effects Threshold (AET) (see Chapter 10)], or in combination with other sediment assessment techniques (e.g., sediment toxicity bioassays). It is commonly used in three ways to assess impacts to benthic communities and sediment quality:

- To compare test and reference stations, for the purpose of determining the spatial extent and magnitude of such impacts

- To identify spatial gradients of impacts

- To identify temporal trends at the same locations through time.

By definition, benthic communities include all organisms living on or in the bottom substrate. For practical reasons, assessments of benthic community structure in soft sediments usually rely on the macrofauna (i.e., organisms retained on a 1.0- or 0.5-mm sieve) and to a lesser extent the meiofauna (i.e., multicellular organisms that pass through a 1.0- or 0.5-mm sieve). Reasons for the more limited use of meiofauna are twofold.

- Although they may be sampled quantitatively, their small size makes working with them difficult, and the taxonomy of many of the groups (e.g., nematodes) is not well known.

- The functional attributes of the various meiofaunal taxa are poorly known, and it is therefore difficult to interpret the importance of the presence or absence of the various taxa in relation to environmental quality. (For example, knowledge of meiofaunal taxa that respond positively or negatively to organic enrichment of the sediments is extremely limited.)

Difficulties in quantitatively sampling other size classes of benthic organisms such as the megafauna (i.e., large organisms that are typically measured in centimeters) and the microfauna (i.e., microbes) usually preclude them from consideration in assessments of benthic community structure. Furthermore, although the functional importance of sediment microbes has been studied, their structural and functional characteristics have not been used as indicators of sediment quality.

1.1 Current Use

Assessments of benthic community structure have been used to describe reference conditions, baseline conditions, and the effects of natural and anthropogenic disturbances. Selected examples of current uses of this approach are provided below.

1.1.1 Organic Enrichment--

Pearson and Rosenberg (1978) performed an extensive review of benthic community succession in relation to organic enrichment of marine and estuarine sediments. Based on that review, they developed a generalized model of structural community changes (i.e., numbers of species, abundances, biomass) in relation to organic enrichment, and identified opportunistic and pollution-tolerant species that are indicative of organic enrichment. Concepts developed by Pearson and Rosenberg (1978) have subsequently been used by many investigators to assess the degree of organic enrichment that has occurred in a variety of soft-bottom habitats. For example, Dauer and Conner (1980) assessed the effects of sewage inputs on benthic polychaete populations in a Florida estuary by collecting information on the total number of individuals, total biomass, and average number of species. They compared the sewage-affected site with a reference site, and examined the response of individual species to organic enrichment. In another study in Florida, Grizzle (1984) identified indicator species based on life history responses to organic enrichment and other physicochemical changes. The taxa identified as indicator species in enriched areas were generally characterized by opportunistic life history strategies. Vidakovic (1983) assessed the influence of domestic sewage on the density and distribution of meiofauna in the Northern Adriatic Sea. He concluded that raw domestic sewage did not have a negative influence on the density and distribution of meiofauna, but the nematode/copepod ratio (Parker 1975) indicated that these stations were under stress.

1.1.2 Contamination Due to Toxic Metals and Metalloids--

Rygg (1985a, 1986) assessed benthic community structure in Norwegian fjords where the disposal of mine tailings had resulted in metals contamination of the sediment. His studies showed an inverse relationship between concentrations of metals in the sediment and the species richness and abundance of the benthic macroinvertebrate fauna. Bryan et al. (1987) examined population distributions of the oyster *Ostrea edulis*, the polychaete *Nereis diversicolor*, and the cockle *Cerastoderma edule* in relation to wastes from metals mining in the Fal Estuary. They concluded that the distribution of species is dependent on their ability to tolerate copper and zinc, and on the capabilities of a population to develop a resistance to metals and thereby maintain their original distribution range.

1.1.3 Contamination Due to Toxic Organic Compounds--

Toxic organic compounds are frequently associated with municipal discharges, industrial effluents, and storm drains. These discharges may also result in organic enrichment and contamination by metals or metalloids. The following benthic studies provided evaluations of sediment quality in areas primarily affected by toxic organic compounds:

- Creosote contamination. Tagatz et al. (1983) examined the benthic communities that colonized uncontaminated sediments and sediments contaminated with three different concentrations of creosote (177, 844, and 4,420 ug/g) in field and laboratory aquaria to assess the effects of marine-grade creosote on community structure. Numbers of individuals and numbers of species in field-colonized communities were significantly lower in all three creosote-contaminated sediments than in the controls. In the laboratory-colonized communities only the two higher creosote concentrations had reduced numbers of individuals and species. Distribution of individuals within species was similar for the laboratory and field assemblages of animals.

- Oil contamination. Elmgren et al. (1983) determined that acute effects of the "Tsesis" oil spill were noted after 16 days on both the macrofauna and meiofauna. Initial recovery was noted 2 yr after the spill. However, the authors predicted that complete recovery would require at least 5 yr. Jackson et al. (1989) investigated the effects of spilled oil on the Panamanian coast, and found that shallow subtidal reef corals and the infauna of seagrass beds had experienced extensive mortality. After 1.5 yr, only some of the organisms in areas exposed to the open sea had recovered. Clifton et al. (1984) performed field experiments in Willapa Bay, WA, and found that oil in the sediments modified the burrowing behavior of infaunal benthos.

1.1.4 Dredging and Construction-Related Activities--

Swartz et al. (1980) examined species richness and species abundances just before dredging occurred in Yaquina Bay, OR, and for 2 yr after dredging. Benthic community recolonization was followed from the appearance of opportunistic taxa through their replacement by less tolerant taxa. Rhoads et al. (1978) examined the influence of dredge-spoil disposal on benthic infaunal succession in Long Island Sound by classifying species into groups based on their appearance in a disturbed area. They suggested that the "equilibrium community is less productive than a pioneering stage" and suggested that productivity may be enhanced through managed disturbances.

1.1.5 Natural Disturbances--

Most studies of natural disturbances have assessed the recovery of benthic communities after the disturbance (e.g., large storms and associated wave activity, oxygen depletion, salinity reductions, El Niño). For example, Dobbs and Vozarik (1983) sampled stations before and after Storm David, and observed that the number of species decreased after the storm. They also documented changes in the rank order of the dominant taxa. Santos and Simon (1980) examined defaunation of benthic communities before, during, and after annual hypoxia in Biscane Bay. They documented that recolonization occurs fairly rapidly after the defaunation period. Oscillations in macrobenthic populations in the shallow waters of the Peruvian coast were examined by Tarazona et al. (1988). Fluctuations in density, biomass, species composition, and diversity were attributed to the El Niño of 1982-1983.

Assessment of benthic community structure is also used as a component of other sediment quality assessment tools. Along with sediment chemistry and sediment toxicity bioassays, it is one of three components of the Sediment Quality Triad (see Chapter 9). It is also a component of the Apparent Effects Threshold approach (see Chapter 10).

1.2 Potential Use

To date, benthic community assessments performed to evaluate sediment quality have focused on the relationships between community variables (e.g., numbers of species, total abundance, biomass) and measures of sediment quality (e.g., organic content, concentrations of chemical contaminants). Only for organic enrichment have individual species been identified that are indicative of various degrees of sediment alteration [see for example Pearson and Rosenberg (1978), Word et al. (1977)]. Moreover, for only a very few species has the autecological relationship between organic enrichment of the sediments and an individual species been explored. [For example, Fabrikant (1984) explored the autecology of the bivalve mollusc *Parvilucina tenuisculpta* in relation to organic enrichment of the sediments in the Southern California Bight.] A tremendous potential exists, however, for identifying species that are indicative (by their persistence, enhanced abundance, reduced abundance, or absence) of sediment contaminants at various concentrations. The identification of such taxa will not be simple because of the complex ecological interactions that occur within benthic communities, and because sediments are frequently contaminated with a mixture of chemicals. A first step in this process might be to attempt to identify species or suites of species that could be used to separate the effects of sediment organic enrichment from sediment contamination by toxic substances.

Another potential use of benthic community assessments would be to predict recovery of benthic habitats following the execution of remedial actions at contaminated sites. To date, it has not been possible to use extant benthic community structure to predict recovery because the only model that relates benthic community structure to sediment quality [i.e., the Pearson and Rosenberg (1978) model] is not quantitative. Quantification of this model and the development of quantitative models for other sediment contaminants will be required before benthic community assessments can be used to predict sediment quality. A valuable byproduct of such models would be the ability to predict the capacity of the remediated area to support higher tropic level organisms that forage on benthic organisms, including commercially and recreationally harvested demersal fishes.

2.0 DESCRIPTION

2.1 Description of the Method

An assessment of benthic community structure typically involves a field survey that includes replicated sampling at each station; sorting and identification of the organisms to species or lowest possible taxon; analyses of the numbers of taxa, numbers of individuals, and sometimes biomass in each sample; and identification of the dominant taxa. Results of the field survey are then interpreted in conjunction with other sediment variables (e.g., sediment grain size, total organic carbon) that were collected concurrently with the benthic samples.

2.1.1 Objectives and Assumptions--

The objective of the benthic community structure approach is to identify degraded and potentially degraded sediments by examining the communities of organisms that inhabit those sediments. This empirical approach assumes the following:

- Because benthic infauna are generally sedentary, benthic community structure reflects the chemical and physical environment at the sampling location

- Benthic community structure may be altered in a predictable manner over time and space by chemical or physical disturbances

- The execution of proper data collection and analysis methods can reduce natural variability of benthic infaunal data and enable the detection of trends in sediment quality.

2.1.2 Level of Effort--

The level of effort required to assess benthic community structure is relatively high. Regardless of the analytical methods, a field survey is required to collect the organisms. The sorting and identification process is labor-intensive and generally expensive. Program objectives will determine whether the data analyses are simple or complex.

2.1.2.1 Type of Sampling Required--The type of sampling required to collect benthic organisms is dependent on the objectives of the sampling program and on the area under study. Usually, the objective of a benthic sampling program is to study the characteristics of and the variation in the benthic community that occupies specific sampling stations. In this case, all organisms present in the sediment at that location are sampled together: those that normally reside in the surface few centimeters of sediment and those that normally reside deeper in the sediment (e.g., 5-15 cm below the surface). In some instances, a sampling program may have a different objective. For example, sampling for the Benthic Resources Analysis Technique (BRAT) (Lunz and Kendall 1982) involves collecting box core samples and determining the biomass (and possibly the communities) present in specific sediment strata (i.e., 0-2 cm, 2-5 cm, 5-10 cm, and 10-15 cm below the sediment surface). In that technique, the benthic data are compared with the benthic organisms consumed by bottom-dwelling fish (as determined by gut content analyses of fish captured in the same area) to determine the food value of the benthos.

Characteristics of the area under study also influence the type of sampling. In intertidal or littoral environments where sampling stations can be occupied by walking to the site, samples are usually collected using a hand-held corer. If stations are located in subtidal areas, then remote sampling from a vessel is performed using a box corer or grab sampler. Sediment grain size may influence final selection of the sampler. Some samplers (i.e., many box corers) perform poorly in sandy sediments while others (i.e., van Veen grab, Smith-McIntyre grab) perform adequately in a greater range of sediment types (i.e., fine to medium sand, silt, silty clay). Methods and equipment for sampling infaunal communities are further described in several publications (Word 1976; Swartz 1978; Eleftheriou and Holme 1984; Nalepa et al. 1988).

Program objectives and knowledge of benthic communities in the study area will influence selection of the sieve size through which sediment samples will be washed. It is important that the sieve mesh sizes be appropriate for the community under study (e.g., 64 um for meiofauna, 0.5 or 1.0 mm for macrofauna). Generally, the chances of retaining most macrofauna species and individuals (and therefore increasing sampling accuracy) are improved by the use of a finer mesh (but, see Bishop and Hartley 1986). However, sieve size is an important determinant of the cost and level of effort necessary to obtain quantitative data. Very little difference in the field processing time exists between use of a 0.5-mm and a 1.0-mm sieve when sieving sediments finer than coarse sand, but laboratory analyses are much more time-consuming when the smaller mesh is used because it retains more abiotic materials and many smaller organisms.

2.1.2.2 Methods--Methods for collecting data on benthic community structure are divided into three categories: program design, field methods, and laboratory methods. Each of these categories is briefly discussed below.

Program design includes the selection of station locations, level of replication, type of sampler, screen size, data analysis methods (discussed later), and quality assurance/quality control (QA/QC) procedures. The selection of station locations will directly influence the usefulness of the resulting data. Stations that will be compared to one another (including reference stations) should be situated in areas with similar hydrography, water depth, and grain size to minimize the natural variability in benthic community composition that can be attributed to these factors. However, such station placement is not always attainable because of altered grain size distributions that often result from contaminant sources.

Selection of the number of replicates is an important component of program design because the accuracy and precision with which benthic community variables are estimated depend in part on the size of the sample (including all replicates). For example, the abundance of a single taxon is generally a less accurate descriptive variable than is the abundance of the total taxa because of the greater variability typically associated with one taxon in comparison with the sum of all taxa. The total area sampled among the replicates at each station should be large enough to estimate a given variable within the limits of accuracy and precision that are acceptable to meet study objectives. A single sample may be useful for general distributional or trends analyses (Cuff and Coleman 1979), but the inherent patchiness of benthic communities makes collection of a sufficient number of replicate samples (a minimum of 3-5, depending on study objectives and sampler area) necessary to ensure statistical reliability (see Elliott 1977). Within a study area, adequate sample size may be determined by maximizing the number of species collected or by minimizing the error associated with the mean for the variable in question (Gonor and Kemp 1978). Additional research on replication is presently being conducted by the U.S. Environmental Protection Agency in Newport, OR under the direction of S. Ferraro (Swartz, R.C., 15 March 1989, personal communication).

Power analysis can assist in determining the appropriate number of replicates. A power analysis includes consideration of the minimum detectable difference in selected biological variables (i.e., the minimum difference in mean values of a variable at several stations that can be detected statistically, given a certain level of variability about those mean values) and the power of the statistical test to be used. The power of the test is especially important because it defines the probability of correctly detecting experimental effects (e.g., differences in biological variables among sampling stations). For a specified variance associated with a biological variable, the statistical power of a test and the minimum detectable difference among sampling areas can be expressed as a function of sample size. The allocation of sampling resources (stations, replication, and frequency) can then be determined with regard to available resources, practicality of design, and desired sensitivity of the subsequent analyses. Discussions and examples of this approach are found in Winer (1971), Saila et al. (1976), Cohen (1977), Moore and McLaughlin (1978), Bros and Cowell (1987), Kronberg (1987), Tetra Tech (1987), Self and Mauritsen (1988), and Vezina (1988).

A potential drawback to use of power analysis is that it requires *a priori* knowledge of variability in the benthic communities that will be studied. If such variability is not known and cannot be estimated, then the number of replicates will probably reflect either funding limitations or generally approved sampling methods. For example, Eleftheriou and Holme (1984) and Swartz (1978) recommend that an area of 0.5 m^2 be sampled to assess species composition in coastal and estuarine regions. Most studies of benthic community structure routinely involve five replicate 0.1-m^2 grab samples. A single 0.1-m^2 grab sample may be sufficient to obtain "useful descriptive information" for use in cluster analyses (Word 1976). However, a single sample precludes direct estimates of within-group variance for statistical analyses. Because individuals are distributed logarithmically among the species of a benthic community (Preston 1948; Sanders 1968; Gray and Mirza 1979), species collected in the second and successive replicates that were not collected in any of the previous replicates most often will be numerically "rare." Note that "rare" is not synonymous with "unimportant." Hence, a single 0.1-m^2 sample is generally not adequate to characterize benthic community structure and function. In general, five 0.1-m^2 grab samples are recommended for determining benthic community structure, unless evaluation of site-specific data (i.e., a power analysis) indicates that sufficient sensitivity can be obtained with fewer samples, or that a greater number is required due to extreme spatial heterogeneity. (Note that at least three samples are required for parametric statistical analyses.)

Another aspect of program design is selection of the appropriate degree of navigational accuracy. For baseline or distributional studies, repeatable station location may not be a high priority, and methods such as Loran C may be sufficient. However, for monitoring programs where reoccupation of exact stations is important (e.g., disposal site monitoring), a more accurate positioning method (e.g., an electronic distance-measuring device or Mini-Ranger) may be required.

A quantitative sampling device and an appropriate mesh size must be selected to ensure that size classes of organisms appropriate for assessing sediment quality are collected. Selection of a sampler and sieve are discussed above, in Section 2.1.2.1.

Field and laboratory methods must be conducted according to rigorous QA/QC protocols. Field methods include collecting, sieving, and preserving the samples. Samples are typically preserved in a solution of 10 percent buffered formalin for at least 24 h. Laboratory methods include rinsing the formalin solution from the samples within 7-10 days, followed by storage in 70 percent ethanol. Samples are sorted under a dissecting microscope during which all organisms are removed from the samples and placed in vials for identification and enumeration of individual taxa. The time required to sort and identify a benthic sample varies greatly depending on the sieve size, sample area, and sediment composition. Sorting may take as little as

1 h for a 0.1-m^2 sample sieved through a 1.0-mm screen, or as much as 12 h if wood chips or other debris are present. The time needed to identify organisms in a sample depends on the number of organisms (which is a function of sieve size, habitat, or degree of contamination) and number of taxa present. The number of hours needed to identify organisms in a sample may range from 1 to over 10 h.

In addition to the collection of samples for analysis of benthic community structure, separate sediment samples should be collected at all stations for conventional sediment chemistry variables (e.g., sediment organic content, sediment grain size distribution). Because organic carbon content and sediment grain size naturally affect the composition of benthic communities, measurement of these variables will assist in determining whether benthic communities are affected by reduced sediment quality.

2.1.2.3 Types of Data Required--The two primary structural attributes of any benthic community are the distributions of species and individuals in three dimensional space, and the distribution of individuals among species and higher taxa. Given an understanding of these two structural attributes, it is possible to infer functional attributes of the benthic community, including trophic relationships, primary and secondary productivity, and interactions between the resident biota and the abiotic habitat. The data required for analysis of the structural and functional attributes include the number of taxa (identifications should be to the lowest taxonomic level possible), the abundance of each taxon, biomass (depending on program objectives), and conventional sediment chemistry variables. However, collection of the appropriate data does not ensure proper evaluation of the structural and functional attributes. The selection and implementation of data analyses are equally important, and are discussed in the remainder of this section. The data analyses presented in this section address primarily structural components of benthic communities. However, functional attributes can be inferred from many of those structural attributes.

Various types of data analyses are used to describe benthic community structure, depending on the objectives of the particular program. However, several descriptive values are common to most program objectives. All organisms collected in each sample are enumerated (i.e., total abundance), and abundances of major taxonomic groups are usually summarized. Depending on the level of identification, abundances of individual taxa, numbers of taxa, and lists and abundances of pollution-tolerant and pollution-sensitive taxa in each sample may be developed. Biomass of major taxonomic groups and total biomass are sometimes reported. The composition of the numerically dominant taxa are analyzed when species level identifications are performed. In addition, descriptive indices such as diversity [the distribution of individuals among species; see Washington (1984) for additional definitions of diversity], evenness (the evenness with which individuals are distributed among taxa), and dominance (the degree to which one or a few species dominate the community) are usually calculated.

Most programs evaluate the temporal or spatial differences in benthic community structure. Typically, comparisons of one or more indices are made at the same station over time and compared to a baseline value, or comparisons are made between stations in a study area and stations in a reference area. If an adequate number of samples is collected (i.e., three or more), statistical tests such as t-tests or Analysis of Variance (ANOVA) (or their nonparametric analogues) are often performed to determine whether significant spatial or temporal differences exist among benthic communities.

Besides univariate (i.e., single variable) statistical analyses, multivariate (i.e., multiple variables) analyses are frequently performed (e.g., Boesch 1977; Green and Vascotto 1978; Gauch 1982; Shin 1982; Long and Lewis 1987; Ibanez and Dauvin 1988; Nemec and Brinkhurst 1988a,b; Stephenson and Mackie 1988). Multivariate analyses include classification (i.e., grouping similar stations into clusters) and ordination [i.e., representing sample or species relationships as faithfully as possible in a low-dimensional (two-four dimensions) space] methods (see Gauch 1982 for an overview of multivariate methods). Multivariate techniques group data and display them on a two-dimensional plot or dendrogram so that stations exhibiting similar communities are located closer to one another than to stations with dissimilar communities. The numerical and graphical results can then be compared with physical and chemical data collected concurrently to determine whether those variables correlate with trends in benthic communities. A commonly used classification technique involves first computing a matrix of similarity indices that represent the degree of similarity in species composition between two stations. Commonly used similarity indices include Bray-Curtis, Canberra metric, and Euclidian distance indices. The similarity matrix is then entered into a clustering algorithm (e.g., pair-wise averaging, flexible sorting) to produce a dendrogram depicting similarities among stations. Commonly used ordination techniques include principal components analysis, detrended correspondence analysis, and discriminant function analysis. Bernstein and Smith (1986) developed an index of benthic community change along pollution gradients that is derived from results of ordination analysis. The index (called Index 5) is a measure of change from reference conditions.

Benthic community surveys generate large data matrices. These data matrices are often reduced by the elimination of certain species (Boesch 1977) prior to performing multivariate analyses. A variety of methods exist for reducing data matrices (see Stephenson et al. 1970, 1972, 1974; Day et al. 1971; Clifford and Stephenson 1975).

Both parametric statistical tests and multivariate analyses may involve data transformations. Transformations of the original data may be necessary for one or more of the following reasons:

- Benthic data sets are usually characterized by large abundances of a few species and small abundances of many species

- The distribution of individuals among species tends to be log-normal

- Sampling effort may be inconsistent (Boesch 1977).

The two basic types of transformations are strict transformations and standardizations. Strict transformations are alterations of the original values (e.g., species abundances) without reference to the range of values within the data. Commonly used transformations are square root, logarithmic, and arcsine (Sokal and Rohlf 1981). Standardizations are alterations that depend on some property of the data under consideration. A common standardization is the conversion of values to percentages.

Benthic data are transformed to better meet the assumptions of parametric tests (e.g., normality, homogeneity of variances). In multivariate analyses, data are often transformed using logarithms [e.g., log (x+1)] because of the presence of zero scores. This transformation is also applied when population variance estimates are positively correlated with mean values (Sokal and Rohlf 1981). Clifford and Stephenson (1975) discuss in detail the effects of transformations on commonly used resemblance measures.

Benthic community structure is usually compared with chemical and physical data that are collected concurrently. These comparisons may take the form of simple linear correlations, correlations with cluster groups, or correlations using multivariate techniques such as discriminant analyses. Multiple discriminant analysis attempts to isolate groups of similar stations so that variables responsible for the separation of groups can be identified. Results may be used to determine whether differences in community structure are due to variations in sediment grain size, variations in other physical characteristics of the environment, or changes in sediment quality due to toxic substances or organic materials.

The use of different methods and analyses may result in different interpretations of the same data. For example, use of the same data with different standardization methods in a classification analysis can yield very different results (Austin and Grieg-Smith 1968). Generally, the more analyses that are conducted on the data, the higher the probability of interpreting the data accurately.

2.1.2.4 Necessary Hardware and Skills--The hardware needed to perform a benthic community assessment is fairly common and should be readily available. Equipment includes field collection gear (e.g., sampling vessel, appropriate sampler, sieves, sample storage containers, buffered fixative); and standard biological laboratory equipment (e.g., microscopes, sieves, hydrometers or pipets, and a balance). More specialized equipment includes a muffle furnace for determining total volatile solids concentrations, a taxonomic reference collection, and a taxonomic reference library. Computer equipment and appropriate software are required to make studies cost-effective. A microcomputer is sufficient for most analyses, but some complicated multivariate analyses may require the use of a mini- or mainframe computer.

Trained benthic taxonomists are required to ensure accurate identifications. Some computer programming and some level of data management are usually required. A trained benthic ecologist is required to synthesize and interpret the data. However, the amount of training depends on the required level of interpretation. For example, interpretation of several multivariate methods would require a higher level of training than interpretation of descriptive indices.

2.1.3 Adequacy of Documentation--

Many different approaches and methods are used to analyze benthic data, some of which have their origins in classical terrestrial community ecology. Because analysis of benthic community structure is a relatively old assessment tool, literally thousands of papers have been written about the method. Several books and protocols have also been developed to describe field and laboratory techniques [e.g., Holme and McIntyre (1984), Puget Sound Protocols (Tetra Tech 1986b), U.S. EPA 301(h) protocols (Tetra Tech 1986a)]. However, a comprehensive document that describes standardized procedures for analyzing and interpreting benthic community data is lacking.

The most commonly used interpretive approaches include measures of diversity and classification. Sometimes a general consensus exists on the best techniques to use within an approach (e.g., widespread use of Shannon-Wiener diversity index, although there is debate as to whether this is a suitable index for environmental impact analysis). Despite this consensus, studies do not necessarily follow a specified format. Program objectives tend to dictate the types of hypotheses posed and analyses used. Many relatively new and exciting approaches have been proposed for assessing benthic community structure. However, most are relatively untested and are not widely used [e.g., benthic resource analysis technique (Lunz and Kendall 1982), abundance-biomass comparison (Warwick 1986; Warwick et al. 1987), infaunal trophic index (Word 1978, 1980), nematode:copepod ratio (Amjad and Gray 1983; Lambshead 1984; Shiells and Anderson 1985; Raffaelli 1987), lognormal distribution (Gray and Mirza 1979), Index 5 (Bernstein and Smith 1986)]. Each of these methods has shown promise in some situations, but more testing and validation are needed before any can gain universal acceptance.

Very few assessments of the information gained from analyses of data at the species level vs. the major taxa level have been undertaken. Warwick (1988) evaluated the results of ordinations run on various hierarchical levels of taxonomic data for five data sets. Three of the data sets were of macrofauna (from Loch Linne, Clyde Sea, and Bay of Morlaix), one was of nematodes from the Clyde Sea, and the last was of copepods from Oslofjord that were subjected to different levels of particulate organic material. He reported that in none of those five cases was there any substantial loss of information at the family level, and that in two cases the sample groupings related more closely to the gradient of pollution at the phylum level than at the species level. Warwick tentatively suggested that "anthropogenic effects modify community composition at a higher taxonomic level than natural environmental variables, which influence the fauna more by species replacement." Warwick's paper appears to be the only published work to support the use of higher taxonomic groups for analysis purposes. In cases where only major taxa level data have been collected (e.g., PTI and Tetra Tech 1988), it has been difficult to determine differences in community structure between impacted areas and reference areas, and to establish causes of community alterations. Although it would be a cost-saving approach, use of higher taxonomic levels to assess benthic communities is currently not an accepted approach in the U. S.

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

The assessment of benthic community structure is directly applicable to the protection of aquatic life. Because benthic organisms are aquatic, assessments of benthic community structure provide a direct measure of the condition of aquatic life. Furthermore, because benthic organisms are consumed by other aquatic organisms (e.g., fish), assessing the condition of benthic communities provides information on other aquatic organisms.

Assessment of benthic community structure is both directly applicable to the protection of some wildlife (e.g., wading shorebirds that feed on the benthic infauna) and indirectly applicable to the protection of other wildlife (e.g., fish-eating wildlife). A substantial decrease in abundance of benthic organisms may result in the loss of food and a reduction in the value of certain habitat to wildlife. For example, distributions of demersal fishes have been shown to be affected by changes in the composition of benthic infaunal communities (e.g., see Kleppel et al. 1980), as has the distribution of the starfish *Astropecten verilli* (Striplin 1987).

Assessment of benthic community structure may be directly or indirectly applied to the protection of human health. When changes in community structure are caused by the presence of toxic contaminants, then the bioaccumulation of those contaminants in more tolerant species may sometimes be postulated. Those contaminated benthic infauna may directly affect human health if they are ingested (e.g., shellfish contamination), or may indirectly affect human health if contaminants are transferred through the food web to humans (e.g., consumption of contaminated demersal fish).

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

Benthic community structure as a stand-alone assessment method cannot presently generate numerical criteria for specific chemicals, nor is it likely that it will without extensive research. However, it is an integral component of other methods that generate numerical criteria (e.g., Apparent Effects Threshold, Sediment Quality Triad). The great number of factors influencing benthic community structure at a given site generally precludes isolation of chemical-specific effects.

3.0 USEFULNESS

Assessment of benthic community structure has become a valued tool for determining sediment quality. It is recognized as the only *in situ* measure that provides information on changes in ecological relationships among species that inhabit potentially contaminated sediment. Its usefulness will continue both as an assessment method on its own, and as a component of other sediment quality assessment tools.

3.1 Environmental Applicability

This method is applicable in a variety of environments. As a tool for assessing sediment quality, it has been used to assess the effects of known or suspected contaminants (e.g., industrial or municipal discharges, oil spills). The results of such studies reveal the geographic extent of the problem area and the type and severity of contamination.

3.1.1 Suitability for Different Sediment Types--

Benthic community structure is well-suited for assessing spatial and temporal effects of chemical contamination and/or organic enrichment in a variety of sediment types. However, to the extent possible, benthic communities occupying different types of sediment should not be compared. Considerable research has shown that the structure of benthic communities in coarse sediments differs from that in fine sediments (see Rhoads and Young 1970; Rhoads and Boyer 1982). Briefly, species recruiting into soft silty sediments must be able to tolerate the deposition of fine particulate material. These environments tend to be inhabited by subsurface deposit-feeding organisms, whereas sandy environments tend to be inhabited by both surface suspension-feeding species and subsurface dwelling species. Therefore, the experimental design of a benthic survey must reflect that the functional attributes of benthic communities in silty and sandy environments fundamentally differ.

When reference stations are used as the basis for determining differences in community structure between nonimpacted and potentially impacted stations, the reference and test stations should exhibit, to the extent possible, similar sediment characteristics (as well as similar water depths because benthic communities naturally vary by depth). However, it is not always possible for the reference and test stations to have sediment that has a similar composition (e.g., dredged material at a dump site may have different characteristics than native sediment surrounding the dump site).

If the experimental design is based upon sampling the same stations through time to assess temporal change, then presumably sediment grain size would remain constant. If the objective is to sample along a potential gradient of chemical contamination or organic enrichment, then all stations should have similar grain sizes and water depths. However, this is not always possible because the source of contamination may alter the natural grain size distribution of the sediments.

Benthic community structure is also a suitable assessment technique for assessing the presence of anaerobic sediments caused by poor flushing or excessive organic loading. The success of this approach will once again hinge on comparing benthic community structure between stations with similar grain sizes and water depths.

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

Analysis of benthic community structure is frequently used to determine effects of chemicals present in the sediment. However, it is not used as a method to quantify the relative concentrations of individual chemicals or classes of chemicals present in sediment. Although individual species may react to certain chemicals, these reactions are not quantifiable at the community level. The Apparent Effects Threshold approach (Chapter 10) incorporates changes in abundance of major taxa for specific chemicals.

Benthic communities respond predictably to general categories of contamination. For example, metals contamination of sediments results in decreased species diversity (Rygg 1985a,b, 1986). Organic enrichment, which leads to an increase in the food supply, generally results in increased diversity and abundance at slight to moderate levels of enrichment (Pearson and Rosenberg 1978; Rygg 1986). However, beyond some level of organic enrichment, diversity and abundance decrease with continued organic loading (Pearson and Rosenberg 1978). In an area receiving both organic enrichment and toxic contaminants, it may be difficult to distinguish the effects of these forms of pollution from each other. Additional research is greatly needed to help separate the effects of multiple sources of contaminants.

3.1.3 Suitability for Predicting Effects on Different Organisms--

Changes in benthic communities that result from the presence of organic enrichment or chemical pollutants may be useful indicators of the potential effects of that pollution on predators of the infauna (see Kleppel 1982; Striplin 1987). However, using benthic community structure to predict specific effects on potential predators (such as benthic-feeding fish or shorebirds) may be difficult. Information on trophic relationships, competition, and predation is often not available. The capability to predict the effects of altered prey communities on predators may improve with research on these topics. Factors such as food quality, distribution of food, interactions among species, and distribution of prey will all be important components of this research.

3.1.4 Suitability for In-Place Pollutant Control--

Benthic community structure has not been used to set sediment quality goals or criteria for polluted marine sediments. Benthic communities naturally express sufficient spatial and temporal variability to eliminate them from consideration as a goal or criterion-setting variable. However, benthic communities are an integral part of other approaches to assess sediment quality (see Chapters 9, and 10, and 11) in which benthic community structure is the only *in situ* biological measure.

3.1.5 Suitability for Source Control--

Benthic community assessments can provide valuable information for certain aspects of source control. Benthic communities can assist the identification of outfalls that discharge toxic chemicals or high organic loads. Depending on the nature of the material being discharged, benthic communities may be diverse and abundant if the material is organically enriched or may be depauperate if the material has high levels of toxic contaminants. Because benthic communities are not currently useful for identifying specific chemicals or classes of chemicals present in the sediment, they are of limited value for identifying specific sources of contaminants.

Following the control of sources, benthic community structure may be used to monitor long-term recovery of the receiving environment (Tetra Tech 1988). It is not recommended as an indicator of the immediate effects of controlling sources because the sediment will remain contaminated until the sediment is actively remediated, or until bioturbation and natural deposition of uncontaminated particulates dilute the contaminated sediment. Furthermore, this assessment technique would be useful only in areas where other uncontrolled sources would not obscure sediment recovery due to the controlled source. Where source control has occurred, or is planned on a regional level, establishment of one or more stations for the analysis of long-term trends in benthic community structure is recommended as a method of monitoring regional sediment recovery. The concentration and type of the contaminants, and hydrodynamics of the study area will govern the length of time over which recovery will occur (Perez, K., 1 May 1989, personal communication).

3.1.6 Suitability for Disposal Applications--

Regulations concerning biological testing of sediment that is dredged under Sections 401 and 404 of the Clean Water Act do not include assessments of benthic community structure. Benthic communities inhabit only the upper layers of sediment that will be dredged. Because sediment quality near the sediment surface may not reflect sediment quality throughout the depth of sediment to be dredged, benthic communities are unable to provide information that is suitable for assessing the entire volume of sediment that will be dredged. Chemical analyses, laboratory bioassays, and bioaccumulation studies can, however, be used to assess sediment quality throughout the dredging depth. Section 102 of the Marine Protection Research and Sanctuary Act does call for monitoring of benthic community structure in areas where dredged material is disposed.

The International Joint Commission (IJC) recommends use of benthic communities to determine whether areas of concern exist in sediments that require dredging (IJC 1988a,b). However, they do not discuss whether benthic community structure would be used to determine the suitability of dredged material for open-water disposal.

Analysis of benthic community structure is appropriate for post-disposal monitoring of confined and unconfined disposal sites and for monitoring recovery of areas that were dredged. As part of the Puget Sound Dredged Disposal Analysis (PSDDA) post-disposal monitoring program, benthic community structure will be used to monitor the potential transport of disposed material away from the disposal site (Tilley et al. 1988). The purpose of this aspect of the monitoring program is to determine whether benthic communities are altered near the disposal site, and if so, whether the changes are due to offsite migration of the disposed material. Benthic community structure was also incorporated into the proposed monitoring program for confined aquatic disposal sites to confirm recolonization of the clean sediment cap and to monitor cap integrity at the Commencement Bay Nearshore/Tideflats Superfund site in Tacoma, WA (Tetra Tech 1988). As described earlier, Swartz et al. (1980) documented recovery in Yaquina Bay, OR following dredging. Rhoads et al. (1978) suggested that periodic disturbance such as dredging and disposal may enhance benthic productivity.

3.2 General Advantages and Limitations

General advantages of using benthic community structure to determine sediment quality include its inherent capability to provide an ecological basis for evaluation of sediment quality. It is an empirical rather than a theoretical approach. However, as with most assessment techniques involving field studies, the evaluation of benthic communities is costly and time-consuming. The information gained is often not suitable for specific management decisions because of the lack of numerical management criteria and the method's inability to identify specific chemicals responsible for an impact. However, the technique has been incorporated into other predictive techniques (see Chapters 9, 10, and 11) that provide information more easily used by resource managers.

3.2.1 Ease of Use--

Assessments of benthic community structure require field collections, extensive laboratory work, and data analysis and interpretation by trained benthic ecologists. It is difficult to argue that the method is easy to use, especially in comparison to other methods that rely on established criteria. However, the use of benthic community structure as a sediment quality assessment tool is widely accepted, and trained benthic ecologists are available throughout the country. By using highly experienced individuals to conduct the field, laboratory, and data analysis work, potential problems (such as generating "noisy" data that obscure real trends, or arriving at different interpretations using the same data) should not occur.

3.2.2 Relative Cost--

The relative cost of conducting an assessment of benthic communities is less than the cost to develop and implement other sediment quality assessment techniques such as the Apparent Effects Threshold and equilibrium partitioning approaches. However, once sediment quality values have been generated, then the relative cost of conducting a benthic survey is greater than the cost of analyzing sediment for contaminant concentrations and comparing those data to the values to determine sediment quality. Sediment toxicity bioassays are generally less costly than analysis of replicated benthic samples. Because the Triad approach requires synoptic analyses of sediment chemistry, sediment toxicity, and benthic communities, it is more costly to implement than simply an analysis of benthic communities. It also provides broader information from which to determine sediment quality.

The objectives of benthic community assessment programs strongly influence cost by dictating the number of stations and number of replicates per station. The cost per replicate is relatively high (i.e., $400-$1,000), but varies greatly depending on the size of the area sampled, the screen size, the level of the taxonomic identifications, and the environment sampled.

3.2.3 Tendency to be Conservative--

Benthic community structure is a moderately conservative measure of sediment quality. Because benthic community structure reflects the collective response of all species, responses of individual species that are susceptible to degradation in sediment quality may not be obvious at the community level because of the lack of response in other species that are more tolerant of environmental degradation. Changes to numerous species or dominant species must occur before changes at the community level are evident. If assessments of sediment quality were made using individual species instead of communities, they could be either conservative by relying on sensitive species, or not conservative by relying on tolerant species.

3.2.4 Level of Acceptance--

Benthic community assessments have been used as a sediment quality assessment tool for several decades in North America, Europe, and Australia, as well as in South Africa, China, and Japan. The method has gained widespread acceptance because of its inherent capability to assess sediment quality at the community level, thereby documenting ecological response to sediment perturbations.

Many methods may be used to analyze benthic community data, as discussed above. Some of these methods have gained far wider acceptance than have other, sometimes newer, approaches. The most widely accepted types of analyses include measures of abundance, numbers of taxa, diversity, similarity, community classification, and the abundance of sensitive and tolerant species. Other analytical methods include the log-normal distribution (Gray and Mizra 1979), the use of major taxa instead of species level data (Warwick 1988), and the Infaunal Trophic Index (Word 1978, 1980). Each of these may be appropriate for certain types of perturbations, but have yet to gain widespread acceptance.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

Many laboratories either have the essential equipment for conducting benthic community surveys, or can readily obtain this equipment. However, locating qualified taxonomists to oversee the sorting and to identify the organisms may be difficult. Taxonomists require several years of training and experience before they are considered experts in their respective taxonomic fields. They also require access to a reference museum of verified organisms to assist in their identifications. A thorough taxonomic library containing original descriptions of species is also an integral component of taxonomic laboratories.

3.2.6 Level of Effort Required to Generate Results--

The level of effort required to conduct a benthic community survey is dependent on the objectives of the program, which may affect the number of stations, number of replicates per station, taxonomic level of the identifications, and data analysis procedures. Regardless of those objectives, a field effort is required, the samples must be sorted, identified, and enumerated, and the resulting data must be analyzed. This process typically requires several months, but it is not unusual for it to require a full year for a very large sampling effort, or for a program in which the samples require large sorting or identification times. For example, the sorting time for samples collected from deep water silt and clay may be 1-2 h, whereas that for samples from shallow sandy sites might require 4-6 h because shallow sandy areas typically contain more abiotic material. If wood chips are present in the sample, then the sorting time can easily exceed 12 h, depending on the volume of wood chips.

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

The interpretation of benthic community data requires an expert who is familiar with the natural history of the fauna and the statistical techniques that are routinely used to analyze the data. Interpretation of the many data points generated by this approach may require many weeks before meaningful trends are recognized. The inherent variability of benthic communities has so far prevented the development of specific benthic criteria for use in assessing pollutant-related trends in sediment quality.

3.2.8 Degree of Environmental Applicability--

The assessment of benthic community structure is a direct measure of the environmental effects of pollutants and, as such, is highly applicable as a method to assess sediment quality. Its applicability lies in its ability to provide information on the effects of pollutants on ecological processes within the sedimentary environment.

3.2.9 Degree of Accuracy and Precision--

Provided that sufficient funding is available to collect and process the necessary numbers of replicate samples, analysis of benthic community structure is accurate (defined as how well the data represent true field conditions) and precise (defined as the consistency and reliability of the samples). The resulting data are obtained directly from the populations under study. Other sediment quality assessment methods described in this compendium are not direct measures of field conditions, and therefore are less likely to be as accurate and precise.

Many factors in the design of a benthic community survey directly influence the degree of accuracy and precision of the resulting data. These factors include station placement, number of replicates, appropriateness of reference areas, sampler, sieve mesh size, sampling interval, quality of taxonomy, and the type and quality of the data analysis. The best way to ensure high degrees of accuracy and precision is to conduct a pilot study in the area of interest prior to designing a major field survey. The pilot survey will provide information on variability within benthic communities, which then directly affects the required number of replicates and station placement. The analysis of data from a pilot study may also help generate different hypotheses that may alter the sampling and analysis plans to better define the communities.

4.0 STATUS

Many methods to assess sediment quality rely on benthic community structure as a measure of potential ecological effects of pollutants. Benthic community structure has been incorporated into programs with vastly different objectives because the resident biota are sensitive indicators of many kinds of environmental perturbations. Aspects of the status of benthic community structure as a sediment quality assessment tool are discussed in this section.

4.1 Extent of Use

Assessment of benthic community structure has been a valued tool in marine, estuarine, and freshwater environments for several decades. Many of the early programs examined benthic communities from an academic viewpoint. Since the 1970s, benthic community structure has been used as a measure of sediment quality. Since then this method has been used to determine the effects of municipal effluents, industrial discharges, eutrophication, organic enrichment, oil spills, and mine tailings disposal (see Section 1.1). It has also been used to determine the suitability of sediments for dredged material disposal, to monitor dredged material disposal sites, and to monitor recovery of impacted areas following the cessation of contaminant loading.

4.2 Extent to Which Approach Has Been Field-Validated

Because benthic community structure is an *in situ* sediment quality assessment tool, it does not require additional field validation.

4.3 Reasons for Limited Use

Although conducting studies of benthic community structure is a common practice, the cost and amount of time required to generate usable results may prevent the method from being implemented by all who could benefit from its use. In fact, the method has been deleted from some programs due solely to cost (Bilyard 1987). In some situations, costs and time have been reduced by taking the identifications only to the major taxonomic level. This reduction of taxonomic detail frequently reduces the usefulness of the information (Warwick 1988), which exacerbates a perception by some resource managers that the data are too variable to be useful. Detecting trends within benthic data is not a simple process. However, the proper design and implementation of a field survey will radically increase the probability of producing valuable data and results.

4.4 Outlook for Future Use and Amount of Development Yet Needed

The outlook for the future use of benthic community structure as a sediment quality assessment tool is particularly bright because of the continuing development of new data analysis methods by researchers in North America and Europe. The objective of these methods is generally to reduce cost or variability within the data by relating aspects of the distributions of organisms or organism biomass to specific kinds of environmental perturbations. Gray and Mirza (1979) determined that the log-normal distribution of individuals was altered in a predictable manner in the presence of slight organic pollution. A more recent method for detecting pollution effects on marine benthic communities is the species abundance/biomass comparison (ABC) method developed by Warwick (1986). This method proposes that the relationship between the number of individuals among species and the distribution of biomass among species changes in a predictable manner in the presence of organic pollution. Beukema (1988) evaluated the ABC method in an intertidal habitat in the Dutch Wadden Sea and determined that the method "cannot be applied to tidal flat communities without reference to long-term and spatial series of control samples." Yet another benthic community assessment method that remains under development is the Infaunal Trophic Index proposed by Word (1978, 1980). That method is based on changes in the feeding ecology of benthic infauna in relation to organic enrichment. The Benthic Resource Assessment Technique, developed by Lunz and Kendall (1982), quantifies the effects of changes in benthic communities on fish resources. Although the BRAT technique is not a direct assessment of benthic community structure, it provides important information on the relationships among benthic communities and higher level predators, and describes how those relationships may change in the presence of pollutants.

A radically different approach to interpreting long-term changes in benthic community structure involves use of a sediment profile camera. Rhoads and Germano (1986) developed the REMOTS™ (remote ecological mapping of the seafloor) system. They use a vessel-deployed sediment-profile camera to photograph vertical sections of the sediment. Although REMOTS™ cannot determine the species composition of the benthic community, it can document relationships between organisms and sediment. Rhoads and Germano (1986) characterized the successional stages of benthic communities, and suggest that mapping these stages will permit the detection of changes in benthic communities. When this information is collected as part of a preliminary survey, it can be used to assist in the design of a cost-efficient benthic community survey for obtaining geochemical and biological information.

The sediment profile camera has been used for a variety of other purposes including assessing the relationships between sediment quality and eutrophication (Day et al. 1987; Revelas et al. 1987; Rhoads, D.C., 1 May 1989, personal communication), monitoring the perimeter of dredged material disposal sites (Rhoads, D.C., 1 May 1989, personal communication; Diaz, R.J., 1 May 1989, personal communication), and evaluating the overwintering habitat of blue crabs in Chesapeake Bay (Schaffner and Diaz 1988). With further research, the sediment profile camera may be used for other applications concerning aspects of benthic community structure and sediment quality.

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CHAPTER 9. SEDIMENT QUALITY TRIAD APPROACH

Peter M. Chapman
E.V.S. Consultants Ltd.
195 Pemberton Avenue
North Vancouver, BC
Canada V7P 2R4
(604) 986-4331

The Sediment Quality Triad (Triad) approach is an effects-based approach to describing sediment quality. It typically incorporates measures of sediment chemistry, sediment toxicity, and benthic infauna communities, although other variables can be used. This combination method is both descriptive and numeric. It is most commonly used to describe sediment qualitatively, but has also been used to generate chemical-specific sediment quality criteria (Chapman 1986, in press-a).

1.0 SPECIFIC APPLICATIONS

1.1 Current Use

The Triad approach can be used to determine the extent of pollution-induced degradation of sediments in a non-numerical, multiple-chemical mode (e.g., Chapman et al. 1986, 1987a, 1988, in preparation; Chapman in press-b). It can also be used to determine numerical sediment quality criteria directly (e.g., Chapman 1986, in press-a) and, through manipulations, to determine Apparent Effects Threshold (AET) values (see Chapter 10). Triad has been used in marine coastal waters on the west coast of North America (e.g., Puget Sound, San Francisco Bay, and Vancouver Harbor, Canada), in the Gulf of Mexico, and in freshwater environments including the Great Lakes (Long and Chapman 1985; Chapman et al. 1986, 1987a, 1988, in press-a, in preparation, unpublished). Current uses of the Triad approach are summarized in Table 9-1 and discussed in Section 3.1 (Environmental Applicability).

1.2 Potential Use

The Sediment Quality Triad approach can also be used to meet the following objectives:

- To identify problem areas of sediment contamination where pollution-induced degradation is occurring
- To prioritize and rank degraded areas and their environmental significance
- To predict where such degradation will occur based on levels of contamination and toxicity.

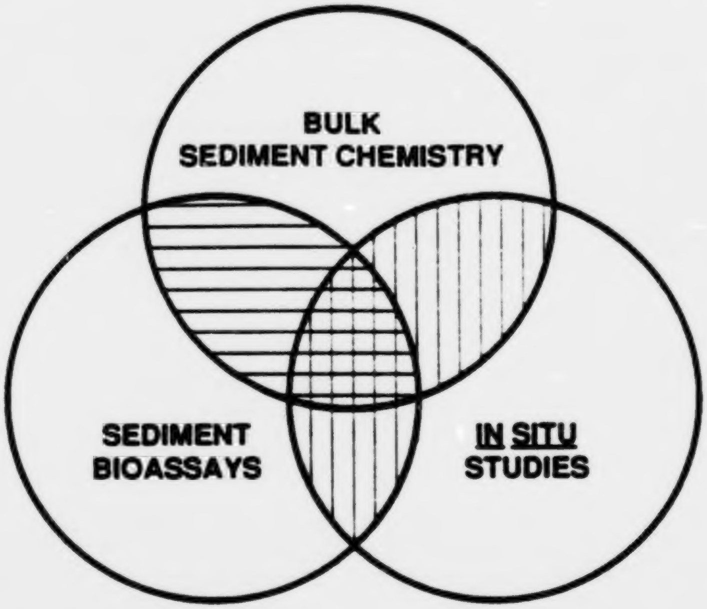
It can be used in any number of situations and is not restricted to aquatic sediments. For example, Triad can be used in water column work with phytoplankton and in terrestrial hazardous waste dump studies with other organisms of concern. Other uses are described in Section 3.1.

2.0 DESCRIPTION

2.1 Description of Method

The Triad approach consists of three components (Figure 9-1):

- Sediment chemistry, to measure chemical contamination
- Sediment bioassays, to measure toxicity



Reference: Adapted from Chapman (1986).

Figure 9-1 Conceptual model of the Sediment Quality Triad, which combines data from chemistry, toxicity bioassays, and *in situ* studies. Chemistry and bioassay estimates are based on laboratory measurements with field collected sediments. *In situ* studies generally include, but are not limited to, measures of benthic community structure. Areas where the three facets of the triad show the greatest overlap (in terms of either positive or negative results) provide the strongest data for determining sediment quality criteria.

TABLE 9-1. CURRENT USES OF THE SEDIMENT QUALITY TRIAD APPROACH

Use	Comment	General Locations Where Implemented[a]
Prioritize areas for remedial actions	Most common usage to date	PS, GM, SF, VH, FW
Determine size of areas for remedial actions	Assuming increasing importance	PS
Verify quality of reference areas	Assuming increasing importance	PS
Determine contaminant concentrations always associated with effects	Common usage; can result in numerical sediment quality criteria and setting of standards	PS
Describe ecological relationships between sediment properties and biota at risk	Along with setting standards and criteria, provides for proactive approach to environmental protection	PS, VH, FW

[a] PS = Puget Sound, various locations (Long and Chapman 1985).
GM = Gulf of Mexico, oil platform (Chapman et al. 1988, in preparation).
SF = San Francisco Bay, various locations (Chapman et al. 1986, 1987a).
VH = Vancouver Harbor, Canada, various locations (Chapman et al. 1989).
FW = Various freshwater environments (Chapman unpublished data; Rogers, North Texas State, unpublished data).

- *In situ* biological variables, to measure *in situ* alteration (e.g., a change in benthic community structure).

The three components provide complementary data. No single component of the Triad approach can be used to predict the measurements of the other components. For instance, sediment chemistry provides information on contamination but not on biological effects. Sediment bioassays provide direct evidence of sediment toxicity. However, the laboratory conditions under which bioassays are conducted that may not accurately reflect field conditions of exposure to toxic chemicals. *In situ* alteration of resident biota measured by infauna community analyses provides direct evidence of contaminant-related effects in the environment, but only if confounding effects not related to pollution (e.g., competition, predation, recruitment cycles, sediment type, salinity, temperature, recent dredging) can be excluded. In particular, because the toxicity of a chemical substance in sediments may vary with its concentration and with the conditions within a specific sediment (e.g., grain size, organic content, pH, Eh, chemical form, presence of other chemicals), the importance of any particular concentration of a chemical or suite of chemicals in sediments cannot be determined solely from chemical measurements.

The three components of the Triad approach integrate chemical and biological response data. They also provide the strong evidence for identifying pollution-induced degradation. For instance, if there are high levels of sediment contamination, toxicity, and biological alteration, the burden of evidence indicates degradation. Conversely, low levels of sediment contamination, toxicity, and biological alteration indicate nondegraded conditions. Conclusions that can be drawn from intermediate responses are listed in Table 9-2.

TABLE 9-2. POSSIBLE CONCLUSIONS PROVIDED BY USING THE SEDIMENT QUALITY TRIAD APPROACH

Possible Outcome	Contamination	Toxicity	Alteration	Possible Conclusions
1.	+	+	+	Strong evidence for pollution-induced degradation
2.	-	-	-	Strong evidence for absence of pollution-induced degradation
3.	+	-	-	Contaminants are not bioavailable
4.	-	+	-	Unmeasured chemicals or conditions exist that have the potential to cause degradation
5.	-	-	+	Alteration is probably not due to toxic chemical contamination
6.	+	+	-	Toxic chemicals are stressing the system
7.	-	+	+	Unmeasured toxic chemicals are causing degradation
8.	+	-	+	Chemicals are not bioavailable or alteration is not due to toxic chemicals

[a] + = Measured difference between test and control or reference conditions.
- = No measurable difference between test and control or reference conditions.

2.1.1 Objectives and Assumptions--

The objectives of the Triad approach are to independently measure sediment contamination, sediment toxicity, and biological alteration, and then use the burden of evidence to assess sediment quality based on all three sets of measurements.

The following assumptions apply:

- The approach allows for 1) the interactions between contaminants in complex sediment mixtures (e.g., additivity, antagonism, synergism), 2) the actions of unidentified toxic chemicals, and 3) the effect of environmental factors that influence biological responses (including toxicant concentrations)

- Selected chemical contaminant concentrations are appropriate indicators of overall chemical contamination

- Bioassay test results and values of selected benthic community structure variables are appropriate indicators of biological effects.

These components are presently treated in an additive manner, with each having equal weight because there is insufficient information available to assign weightings.

2.1.2 Level of Effort--

Ideally, the Triad approach would be based on the use of synoptic data. Sediments for analysis of toxicity should come from the same composited homogenate, as detailed by Chapman (1988), and benthic infauna samples should be collected at the same sampling location. Chemistry and bioassay sediments are collected (usually by remote grab), transferred to a solvent-rinsed glass or stainless steel bowl, and thoroughly homogenized by stirring with a glass or stainless steel spatula until textural and color homogeneity are achieved. The homogenized sediments are then placed in separate sampling containers. In general, chemistry and bioassay samples include laboratory rather than field replication. Benthic infaunal samples are collected at the same location. In the absence of initial sampling to determine the optimum level of replication at a site, five field replicate benthic samples are recommended per station (see Chapter 7, Section 2.1.2.2 herein). Coincident rather than synoptic sampling is possible (e.g., Long and Chapman 1985), but data interpretation is complicated by spatial heterogeneity in sediment contamination and toxicity (cf. Swartz et al. 1982).

Adequate quality assurance/quality control (QA/QC) measures must be followed in all aspects of the study, from field sampling through laboratory analyses and data entry. Detailed QA/QC procedures are available through international (e.g., Keith et al. 1983) and regional publications (e.g., Tetra Tech 1986b).

The first component of Triad involves identification and quantification of inorganic and organic contaminants present in the sediments. Chemical analytes measured are generally restricted by equipment, technology, and the availability of funds and facilities. Local concerns and existing data also affect target analytes measured. Fiscal conservatism, if a factor, must be balanced against the need for a analytical database sufficiently large to allow determination of the presence (or absence) of known toxicants of concern.

An example of the types and classes of compounds required to provide a reasonable characterization of chemical contamination is shown in Table 9-3. Total organic carbon and grain size are measured to provide a basis for normalizing the data to different types of sediments. Coprostanol, an

TABLE 9-3. EXAMPLE ANALYTES AND DETECTION LIMITS FOR USE IN THE CHEMISTRY COMPONENT OF TRIAD

Analyte	Detection Limit	Analyte	Detection Limit
Conventionals (mg/kg, dry)			
Grain size	n/a	Biphenyl	5
TOC[a]	n/a	Perylene	5
Sulfides	0.5	Coprostanol	10
Inorganics (mg/kg, dry)		op'-DDD	0.15
Arsenic	0.05	op'-DDE	0.25
Iron	2.5	op'-DDT	0.15
Chromium	1.0	pp'-DDD	0.15
Copper	0.5	pp'-DDE	0.10
Cadmium	0.05	pp'-DDT	0.10
Lead	0.05	Dieldrin	0.10
Mercury	0.01	Heptachlor	0.10
Nickel	1.0	Hexachlorobenzene	0.10
Silver	0.05	Lindane	0.15
Selenium	0.05	Mirex	0.10
Zinc	0.5	PCBs[c]	2.5
Organics (ug/kg, dry)		PCP[d]	1.0
LPAH[b]	5	TCP[e]	1.0
Benzo(a)pyrene	10		
Benz(e)pyrene	10		
Benz(a)anthracene	10		
Chrysene	10		
Dibenzanthracene	16		
Fluoranthene	5		
Pyrene	5		

The detection limits are the instrumental estimates. Actual detection limits may be higher because of matrix effects.

[a] TOC = total organic carbon.

[b] LPAH = low molecular weight polycyclicaromatic hydrocarbons (includes acenapthene, anthracene, naphthalene and methylated naphthalenes, fluorene, phenanthrene, and methylated phenanthrenes).

[c] PCBs = polychlorinated biphenyls.

[d] PCP = pentachlorophenol.

[e] TCP = tetrachlorophenol.

indicator of human waste, is measured to differentiate sewage inputs from industrial inputs.

The second Triad component involves identification and quantification of toxicity based on laboratory tests using field-collected sediments. Ideally, one would test the toxicity of the sediments to all ecologically and commercially important fauna living in or associated with the sediments. For logistical reasons, a small number of bioassays is conducted to cover as wide a range as possible of organism type, life-cycle, exposure route, and feeding type. The number of tests undertaken is affected by the same constraints as those mentioned for sediment chemistry analyses.

Possible static sediment bioassays that provide a reasonable characterization of the degree of toxicity are shown in Table 9-4. Sediment bioassays with estuarine waters have been developed but are not yet as well accepted as those for fresh and marine waters. Obvious omissions from this list include full life-cycle chronic tests, and genotoxic or cytotoxic response tests. Such tests merit consideration for inclusion when proven accepted methods become available (e.g., Long and Buchman in press).

The final Triad component involves the evaluation of *in situ* biological alteration. Generally this component is provided by benthic infauna community data because benthic organisms are relatively sessile and location-specific. Histopathology of bottom fish has also been used for this Triad component (Chapman 1986), but for area-wide rather than site-specific studies, as these fish are relatively mobile. Several variables in combination are effective in characterizing benthic community structure for the Triad approach: numbers of taxa, numerical dominance, total abundance, and percentage composition of major taxonomic groups. In the marine environment, this last category includes any or all of polychaetes, amphipods, molluscs, and echinoderms. In the freshwater environment, oligochaetes, chironomids, and other major insect groups would fit into the last category.

TABLE 9-4. POSSIBLE STATIC SEDIMENT BIOASSAYS

Bioassay	Duration	Endpoint	Amount of Sediment Required (L)
Marine Waters			
Rhepoxynius abronius (adult amphipod)	10 days	Survival, avoidance	1.5
Bivalve larvae development	48 h	Survival, development	0.5
Neanthes sp. (juvenile polychaetes)	20 days	Survival, growth	2.0
Fresh Waters			
Hyalella azteca (adult amphipod)	10 days	Survival, avoidance	1.5
Daphnia magna (water flea)	10 days	Survival, reproduction	0.5
Chironomus tentans (juvenile insect)	25 days	Survival, growth	1.5
Estuarine Waters			
Eohaustorius estuarinus (adult amphipod)	10 days	Survival, avoidance	1.5

Sediment chemistry, toxicity, and benthic infauna data are combined in the Triad approach to assess the degree of degradation of each station and of each site (Figure 9-1). All data are compared on a quantitative basis, and are normalized to reference site values by converting them to ratio-to-reference (RTR) values as described by Chapman et al. (1986, 1987a) and Chapman (in press-b). The reference site chosen (either *a priori* or *a posteriori)* is generally the least contaminated site of those sampled, and ideally its sediment and other characteristics (e.g., water depth) would be similar to those of the other sites. To determine RTR values, the values of specific variables (e.g., normalized concentration of a particular metal, percent mortality in a particular bioassay, number of taxa) are divided by the corresponding reference values. This process normalizes the data so that they can be compared even when, for instance, there are large differences in the units of measurement. The reference site may be a single station (whose RTR value is 1.0 by definition) or an area containing several stations for which data are averaged.

The RTR criterion is based but not dependent on the assumption that the reference site concentrations are, in fact, indicative of reference or background conditions. The degree to which chemical concentrations are elevated above the mean reference concentrations at a selected site is used as the criterion for selecting chemicals most likely to be anthropogenically enriched and of concern. An index of contamination is calculated for each station by separately determining RTR values for groups of similar chemicals (e.g., metals, PAH, chlorinated organics), and then, assuming additivity, combining these values as a single mean chemistry RTR value. An index of toxicity is calculated by combining bioassay RTR values as a single mean value. An index of biological alteration is calculated in the same manner as is toxicity, using benthic community structure data. The indices of contamination are used to rank stations. These summary ranks are also compared with the ranks generated using the sediment bioassay and infaunal data.

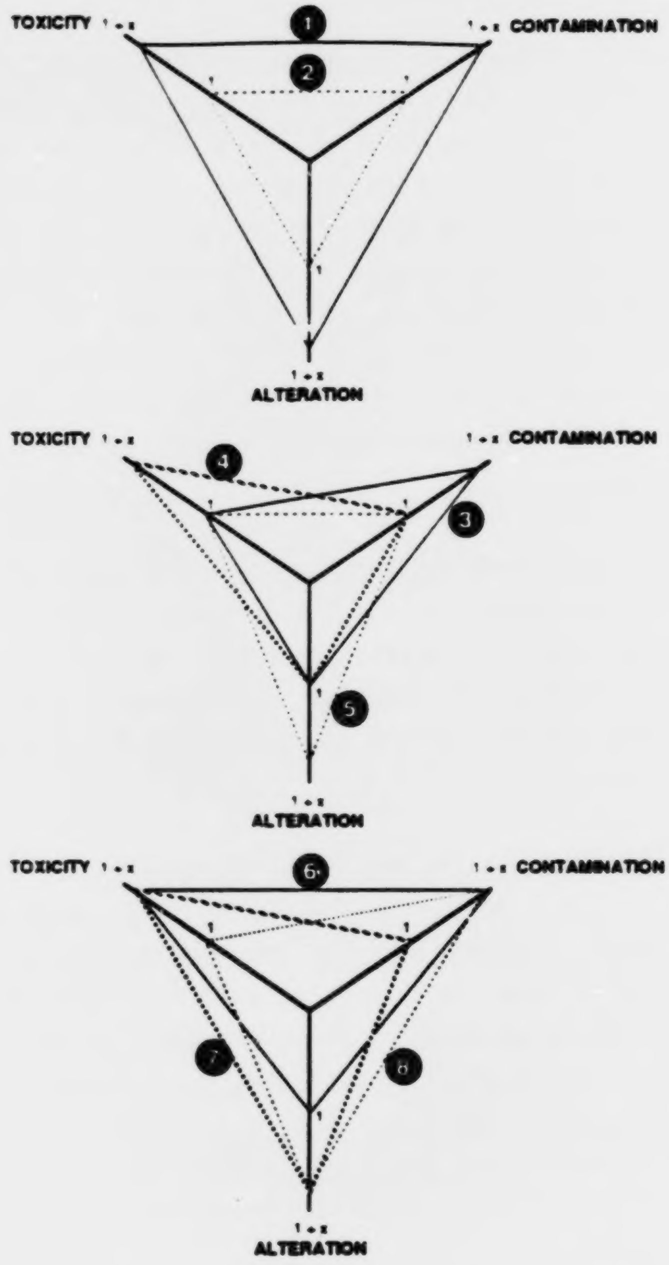
The composite RTR values for each Triad component can also provide useful visual indices. These values can be plotted on scales with a common origin and placed at 120 degrees from each other such that each of the three values becomes the vertex of a triangle. The relative degree of degradation is derived by calculating and comparing the areas of the triangles for each station or site. Examples of such triaxial plots are shown in Figure 9-2, for the eight possible situations detailed in Table 9-2. These plots also provide a visual guide to the characteristics of "background" or reference stations. Since reference data usually involve a site containing more than one reference station, RTR comparisons should also be made against individual reference stations.

2.1.2.1 Type of Sampling Required--Synoptic sampling is preferred for all three Triad components, as described above. Any reasonable sampling procedure can be used if it provides suitable sediment samples for quantifying sediment contamination, toxicity, and biological alteration. Studies to date have used remote samplers such as a 0.1-m^2 van Veen grab operated from a vessel.

2.1.2.2 Methods--Typical variables included in the chemical analyses and sediment bioassays are listed in Tables 9-3 and 9-4, respectively. Details for benthic infauna analyses are provided in Chapter 7 herein. Although unit costs vary, costs are typically on the order of $1,500 for three separate replicated (n=5) sediment bioassays, $1,500 for unreplicated chemical analyses, and $2,000 for replicated (n=5) benthos.

2.1.2.3 Types of Data Required--Standard measurements of chemistry, toxicity, and biological alteration are required. These measurements are then combined, as described above.

2.1.2.4 Necessary Hardware and Skills--Appropriate sampling equipment and trained field and laboratory personnel are required for chemical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7
8
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1 + x
ALTERATION

Reference: Chapman (in press-b).

Figure 9-2. The Sediment Quality Triad determined, in the example situation, for each of the eight possible outcomes described in Table 9-2. Toxicity, contamination, and alteration are shown normalized to Ratio-to-References values as described by Chapman et al. (1986, 1987a), 1.0 = reference conditions. Note that the exact symmetry in these examples would not be routinely expected in actual studies.

analyses, toxicity testing, and benthic infaunal analyses. Although the equipment required can be both costly and sophisticated, it is in common use for investigations related to sediment contamination. The necessary equipment, facilities, and expertise are generally available through a wide variety of government, university, commercial, and private facilities.

2.1.3 Adequacy of Documentation--

Documentation for use of this method is provided by Long and Chapman (1985), Chapman (1986, in press-a, in press-b), and Chapman et al. (1986, 1987a, 1988, in preparation). Using many of the references cited immediately above, other investigators have successfully applied this method (e.g., Wiederholm et al. in press).

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

This approach is directly applicable to the protection of aquatic life. To date, only benthic invertebrates and fish have been used to assess *in situ* biological effects and sediment toxicity. Protection of aquatic life may indirectly protect wildlife (e.g., wading birds feeding on benthos) and humans (e.g., via consumption of aquatic life). The approach can be directly applicable to human health and wildlife protection if the Triad components are redirected towards issues such as bacterial contamination and toxic contaminant bioaccumulation. For instance, Triad could be used to address bacterial problems by 1) measuring bacterial contamination in water or sediment, 2) measuring bacterial diseases or concentrations in tissues, and 3) performing laboratory tests to quantify relationships between sediment/water concentrations and effects. Toxic contaminant bioaccumulation could be addressed by 1) measuring toxic contaminant concentrations in water or sediment, 2) measuring bioconcentration/biomagnification in tissues, and 3) performing laboratory tests to determine effects related to bioconcentration and biomagnification.

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

Triad has been used to generate criteria for three contaminants: lead, PAH, and PCBs (Chapman 1986). These criteria were developed in Puget Sound by examining large data sets to identify areas and concentrations that were associated with no or minimal biological effects. The criteria fall within a factor of 2 to 10 of values generated for these contaminants by the screening level concentration (see Chapter 10, Section 1.1.2), the AET approach (see Chapter 10), and laboratory toxicity methods (Chapman et al. 1987b). As detailed by Chapman (in press-a), the Triad approach is similar to the AET approach except that the former combines all bioassay and *in situ* biological effects data to provide a single value, while the latter provides criteria for benthic infauna and each bioassay conducted. However, there has been little work since Chapman (1986) on development of the Triad approach for the production of numerical sediment quality criteria.

3.0 USEFULNESS

3.1 Environmental Applicability

Although the Triad approach is both labor-intensive and expensive, its strengths render it extremely cost-effective for the level of information provided. First, it provides empirical evidence of sediment quality (i.e., based on observation, not theory). Second, it allows ecological interpretation of physical, chemical, and biological properties (i.e., interpretation of how these relate to the real environment). Third, it uses a preponderance-of-evidence approach rather than relying on single measurements (i.e., all the data are considered). Because of the comprehensive nature of Triad studies, additional follow-up studies are usually not necessary. Finally, the data generated by the Triad approach can be used to generate effects-based classification indices.

The Triad approach enables investigators to estimate the size of degraded and nondegraded areas. It also provides a test of the quality of reference areas (i.e., do contamination or biological effects occur?). Standards in the form of sediment quality criteria (Chapman 1986, in press-a; PTI 1988a,b) can be set from the contaminant concentrations that are always associated with effects. The Triad approach also provides the information necessary to describe the ecological relationships between sediment properties and biota at risk from sediment contamination.

The Triad approach has been used in dredging studies to support dredged material disposal siting and disposal decisions (Chapman unpublished). In multiplying the relative degree of degradation at a site by the volume of sediment to be dredged, investigators can compare different sites, provided that the same reference area is used to develop RTR values. This comparison helps investigators determine whether dredging will affect useful habitat or result in material that is unacceptable for ocean disposal. Similarly, potential disposal sites can be compared with each other and with the material to be dredged, and then compared to acceptability criteria for various uses and options. This application of the Triad approach replaces similar but less useful comparisons based solely on the total mass of chemical contaminants to be dredged.

In areas where benthic communities have been eliminated or drastically changed due to a natural event (e.g., storm, oxygen depletion) or physical anthropogenic impact (e.g., recent dredging, boat scour), the other two Triad components (i.e., sediment chemistry and toxicity) provide information where conventional univariate approaches would prove deficient. Such cases enforce the need to use knowledge of an area in making any type of environmental assessment, including the Sediment Quality Triad.

The Triad approach can be used to discern and ultimately to monitor regional trends in sediment quality. Such information is necessary to delineate areas that are excessively contaminated with toxic chemicals affecting the biota and, therefore, most in need of remedial action. Pilot studies of this nature have been conducted in Puget Sound and San Francisco Bay (Long and Chapman 1985; Chapman 1986; Chapman et al. 1986, 1987a) and in freshwater environments in Europe (e.g., Wiederholm et al. in press).

3.1.1 Suitability for Different Sediment Types--

The Triad approach can be used with all sediment types, including sands, muds, aerobic sediments, and anaerobic sediments. It includes sediment characterization with physical parameters (e.g., grain size, and TOC) that may be important in interpreting the Triad compounds. For example, caution must be used in interpreting the results of toxicity tests in sediments that remain anaerobic in the laboratory despite aeration. Specifically, organisms will die because of lack of oxygen, making it difficult to distinguish that mortality from toxicity due to high concentrations of contaminants.

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

The Triad approach can be used with all chemicals or classes of chemicals, provided that bioassay organisms and tests are appropriate for all different chemicals. For this reason, a battery of bioassay tests is recommended. Caution must be used when testing sediment extracts that may be specific to certain chemical classes. Interpretation of the results must be restricted to only those chemicals.

3.1.3 Suitability for Predicting Effects on Different Organisms--

Application of the Triad approach can be limited by the organisms in the environment if the *in situ* effects are determined primarily by the same species that are used in the bioassay tests. In other words, all biological effects data are based on a single species. In such cases, independence of the infaunal community analyses and bioassay test results cannot be assumed.

Hence, more than one bioassay test is recommended. Ideally, the tests would include a wide variety of organisms, life-stages, feeding types, and exposure routes.

3.1.4 Suitability for In-Place Pollutant Control--

The Triad approach provides a comprehensive approach to in-place pollutant control as it allows for assessment of all potential interactions between chemical mixtures and the environment. The comprehensiveness of this method results from the fact that it includes measurements of multiple chemicals as well as potential toxic effects of both measured and unmeasured chemicals.

3.1.5 Suitability for Source Control--

The Triad approach is as suitable for source control as it is for in-place pollutant control. It can be an environmental complement to toxicity reduction evaluation (TRE) programs that involve chemical and toxicity investigations of effluents and other discharges.

3.1.6 Suitability for Disposal Applications--

The Triad approach has been used for disposal applications, including Navy Homeporting work in San Francisco Bay. In that study, it clearly separated potential dredge sites from one another in terms of the relative level of pollution. Although the Triad was not used in the final decision because of other considerations, decision-makers were able to use information provided by the Triad to compare the suitability of dredging and disposal options.

3.2 General Advantages and Limitations

There are several major advantages to the Triad approach:

- The combination of the three separate components of Triad provides a preponderance-of-evidence approach
- This approach does not require *a priori* assumptions concerning the specific mechanisms of interaction between organisms and toxic contaminants
- This method can be used to develop sediment quality values (including criteria) for any measured contaminant or a combination of contaminants, including both acute and chronic effects
- It provides empirical evidence of sediment quality
- It can be used for any sediment type
- It allows ecological interpretation of both physical-chemical and biological properties
- Follow-up is usually not necessary when a complete Triad study is conducted.

There are also several major limitations to the Triad approach:

- Statistical criteria have not been developed for use with the Triad approach

- Rigorous criteria for calculating single indices from each of the sediment chemistry, bioassay, and *in situ* biological effects data sets have not been developed

- A large database is required

- If this method is used to determine single-chemical criteria, results could be strongly influenced by the presence of unmeasured toxic contaminants that may or may not covary with measured chemicals

- Methods for sediment bioassay testing need to be standardized

- Sample collection, analysis, and interpretation is labor-intensive and expensive

- The choice of a reference site is often made without adequate information regarding how degraded that site may be.

3.2.1 Ease of Use--

The Triad approach is relatively easy to use and understand. The concept is straightforward. A high level of chemical and biological expertise is required to obtain the data for the three separate Triad components. However, many laboratories or groups of laboratories possess the required expertise.

3.2.2 Relative Cost--

Relative cost can be evaluated in terms of either dollars or environmental damage. The Triad approach will not prevent environmental damage, but it can be used to identify contaminated areas for future remediation. In terms of dollars, the Triad approach requires substantial resources to be implemented properly, although step-wise, tiered use of Triad components is possible. Measured against the potential environmental damage due to toxic contamination and the costs of remediation, the Triad approach is not expensive.

3.2.3 Tendency to be Conservative--

The Triad approach provides objective data with which to determine and sometimes to predict environmental damage. Its predictive ability allows for but does not require conservatism on the part of the decision-makers.

3.2.4 Level of Acceptance--

The Triad approach is gaining a high level of acceptance in various parts of North America and in Europe (Forstner et al. 1987; Wiederholm et al. in press). In addition, Canada will be conducting Triad studies in Vancouver and Halifax to determine the suitability of this approach for implementation of the new Canadian Environmental Protection Act.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

All aspects of the Triad approach (i.e., benthic infaunal studies, sediment chemistry analyses, sediment toxicity bioassays) can be conducted by any competent, specialist laboratory that is reasonably well equipped. The major requirements are adequate QA/QC procedures for chemical measurements; appropriate detection limits; and, for biological analyses, taxonomic experts and a taxonomic reference library or museum.

3.2.6 Level of Effort Required to Generate Results--

Different levels of effort will generate different levels of results. For instance, results can be generated by simply measuring one or two chemicals, determining the number of infauna present, and conducting a single sediment toxicity bioassay. However, the applicability of these results may be severely limited. Consequently, multiple chemicals including inorganic and organic compounds should be measured, and multiple measures of *in situ* biological alteration and sediment toxicity should be made. Although it is possible to use previously collected nonsynoptic data to derive results in a "paper" study (e.g., Long and Chapman 1985), fieldwork and synoptic sampling generate the most useful results.

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

Beyond the general conclusions noted in Table 9-2, expert judgment is required to implement and interpret the Triad approach. In particular, the definition of "minimal" and "severe" biological effects is required to establish chemical-specific criteria. The Triad approach reflects the complexity of the issues that must be addressed to assess environmental quality.

3.2.8 Degree of Environmental Applicability--

The Triad approach has an extremely high degree of environmental applicability, as detailed above in Section 3.1.

3.2.9 Degree of Accuracy and Precision--

The accuracy and precision of the Triad approach have not been quantitatively determined. It is expected to have a high degree of accuracy and precision, although these parameters will vary with those of the constituent components.

4.0 STATUS

4.1 Extent of Use

Development of the formalized Triad concept has occurred relatively recently (Long and Chapman 1985; Chapman 1986; Chapman et al. 1986, 1987a, 1988, in press-b). The Triad approach has been used directly to establish sediment quality criteria (Chapman 1986) and, through data manipulations, to determine AET values for sediment quality criteria (Tetra Tech 1986a; PTI 1988a,b).

Triad has been used to identify spatial and temporal trends of pollution-induced degradation. Indices developed using the Triad approach can be numeric (i.e., numeric sediment quality criteria) or primarily descriptive (see Figure 2; Chapman et al. 1987a). In either case, the Triad approach provides an objective identification of sites where contamination is causing discernible harm.

4.2 Extent to Which Approach Has Been Field-Validated

The Triad approach includes field measurements of *in situ* biological alteration. As such, it can be considered that field validation is an integral part of each and every complete Triad investigation.

4.3 Reasons for Limited Use

As previously described, the Triad approach is being used in the U.S., Canada, and Europe for marine, estuarine, and freshwater areas. It is not being used in small projects due to cost and expertise required for full implementation.

4.4 Outlook for Future Use and Amount of Development Yet Needed

The following areas of the Triad approach require development:

- Determining the appropriateness of different endpoints of different bioassays, selected chemical contaminants, selected measures of benthic community structure, and other potential measures of *in situ* biological alteration

- Determining the appropriateness of an additive treatment of the data (e.g., summing bioassay responses to provide a single index for toxicity).

- Development of statistical criteria.

- Development of rigorous criteria for determining single indices for each of the three Triad components.

- Methods standardization for sediment toxicity bioassays.

However, even without development of the above, the Triad approach provides valuable information. The argument has been made (Chapman et al. 1986, 1987a) that the Triad approach provides objective information on which to judge the extent of pollution-induced degradation. For this reason alone, it is expected that the Triad approach will be much more widely used in future.

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10.0 APPARENT EFFECTS THRESHOLD APPROACH

Catherine Krueger
Office of Puget Sound
U.S. Environmental Protection Agency Region X
1200 Sixth Avenue
Seattle, WA 98101
(206) 442-1287

In the Apparent Effects Threshold (AET) approach, empirical data are used to identify concentrations of specific chemicals above which specific biological effects would always be expected. Following the development of AET values for a particular geographic area, they can be used to predict whether statistically significant biological effects are expected at a station with known concentrations of toxic chemicals.

1.0 SPECIFIC APPLICATIONS

1.1 Current Use

At present, the AET approach is being used by several programs to develop guidelines for the protection of aquatic life in Puget Sound. These guidelines are the culmination of cooperative planning and scientific investigations that were initiated by several federal and state agencies in the early and mid-1980s.

Three programs and applications of the AET approach are highlighted below. Notably, all these programs involve an element of direct biological testing in conjunction with the use of AET values, in recognition of the fact that no approach to chemical sediment quality values is 100 percent reliable in predicting adverse biological effects. An underlying strategy in many of these programs was to develop two sets of sediment quality values based primarily on AET values:

- One set of values identifies low chemical concentrations below which biological effects are improbable

- A second set of values identifies higher chemical concentrations above which multiple biological effects are expected.

The programs incorporate direct biological testing in concentration ranges between these two extremes to serve as a "safety net" (i.e., to account for the uncertainty of chemical predictions) for potential adverse effects or anomalous situations at "moderate" chemical concentrations.

1.1.1 Commencement Bay Nearshore/Tideflats Superfund Investigation--

Commencement Bay is a heavily industrialized harbor in Tacoma, WA. Recent surveys have indicated over 281 industrial activities in the nearshore/tideflats area. Comprehensive shoreline surveys have identified more than 400 point and nonpoint source discharges in the study area, consisting primarily of seeps, storm drains, and open channels. A remedial investigation (RI) under Superfund, started in 1983, revealed 25 major sources contributing to sediment contamination, including major chemical manufacturing, pulp mills, shipbuilding and repair, and smelter operations. Adverse biological effects were found in sediments adjacent to these sources.

The AET approach was developed during the course of the RI to assess sediment quality using chemical and biological effects data [i.e., depressions in the number of individual benthic taxa, presence of tumors and other abnormalities in bottom fish, and several laboratory toxicity tests (amphipod mortality, oyster larvae abnormality, bacterial bioluminescence)]. AET values were also used in the subsequent feasibility study (FS) to identify cleanup goals and define volumes of contaminated sediment for remediation.

The AET values used in the FS were generated from a reduced set of biological effects indicators, which comprised depressions in total benthic abundance, amphipod mortality, oyster larvae abnormality, and bacterial luminescence.

1.1.2 Puget Sound Dredged Disposal Analysis Program--

In 1985, the Puget Sound Dredged Disposal Analysis (PSDDA) program was initiated to develop environmentally safe and publicly acceptable options for unconfined, open-water disposal of dredged material. PSDDA is a cooperative program conducted under the direction of the U.S. Army Corps of Engineers (Corps) Seattle District, U.S. EPA Region X, the Washington Department of Ecology (Ecology), and the Washington Department of Natural Resources (WDNR). AET values were used to develop chemical-specific guidelines to determine whether biological testing on contaminated dredged material is needed. Results of the biological testing help determine suitable disposal alternatives.

Above a specified chemical concentration (i.e., the screening level concentration or SLC) biological testing is required to determine the suitability of dredged material for unconfined, open-water disposal. Based primarily on AET values for multiple biological indicators, a higher "maximum level concentration" was also identified. Above this latter concentration, failure of biological tests is considered to be predictable. However, an optional series of biological tests can be conducted under PSDDA to demonstrate the suitability of such contaminated material for unconfined, open-water disposal (Phillips et al. 1988).

1.1.3 Urban Bay Toxics Action Program--

The Urban Bay Toxics Action Program is a multiphase program to control pollution of urban bays in Puget Sound. The program includes steps to identify areas where contaminated sediments are associated with adverse biological effects, specify potential pollution sources, develop an action plan for source control, and form an action team for plan implementation. Initiated in 1984 by Ecology and U.S. EPA Region X's Office of Puget Sound, the program is a major component of the Puget Sound Estuary Program (PSEP). Substantial participation has also been provided by the Puget Sound Water Quality Authority (Authority) and other state agencies and local governments. Major funding and overall guidance for the program is provided by U.S. EPA Office of Marine and Estuarine Protection.

In the PSEP urban bay program, AET values are used in conjunction with site-specific biological tests during the assessment of sediment contamination to define and rank problem areas. Source control actions are well underway, but sediment remediation has not yet begun at any of the sites (PTI 1988).

1.2 Potential Use

The AET approach to determining sediment quality can also be used as follows:

- To determine the spatial extent and relative priority of areas of contaminated sediment

- To identify potential problem chemicals in impacted sediments and, as a result, to focus cleanup activities on potential sources of problem contaminants

- To define and prioritize laboratory studies for determining cause-effect relationships

- With appropriate safety factors or other modifications, to screen sediments in regulatory programs that involve extensive biological testing.

Proposed regulations for sediment contamination are currently under review in Puget Sound. These regulations may include use of AET values to develop statewide sediment quality standards. Ecology is currently developing a suite of sediment management standards, as mandated by the Puget Sound Water Quality Authority (1988) in its 1989 Management Plan. The proposed standards are based in part on AET values. Development of these standards (Becker et al. 1989) relies heavily on the past and ongoing efforts described in Section 1.1 and involves active participation by Ecology, U.S. EPA, the Authority, WDNR, the Corps (Seattle District), and various public interest groups. The draft regulation currently under development affects only sediments in Puget Sound. As additional data become available from other locations, the adopted regulation will eventually be broadened and modified to include the entire state.

2.0 DESCRIPTION

2.1 Description of Method

AET values are derived using a straightforward algorithm that relates biological and chemical data from field-collected samples. For a given data set, the AET for a given chemical is the sediment concentration above which a particular adverse biological effect (e.g., depressions in the total abundance of indigenous benthic infauna) is always statistically significant ($P \leq 0.05$) relative to appropriate reference conditions. The calculation of AET for each chemical and biological indicator is conducted as follows:

1. **Collect "matched" chemical and biological effects data--** Conduct chemical and biological effects testing on subsamples of the same field sample. (To avoid unaccountable losses of benthic organisms, benthic infaunal and chemical analyses are conducted on separate samples collected concurrently at the same location.)

2. Identify "impacted" and "nonimpacted" stations--Statistically test the significance of adverse biological effects relative to suitable reference conditions for each sediment sample. Suitable reference conditions are established by sediments exhibiting very low or undetectable concentrations of any toxic chemicals, an absence of other adverse effects, and physical characteristics that are directly comparable with those of the test sediments.

3. Identify AET using only "nonimpacted" stations--For each chemical, the AET can be identified for a given biological indicator as the highest detected concentration among sediment samples that do not exhibit statistically significant effects. (If the chemical is undetected in all nonimpacted samples, then no AET can be established for that chemical and biological indicator.)

4. Check for preliminary AET--Verify that statistically significant biological effects are observed at a chemical concentration higher than the AET; otherwise the AET should be regarded only as a preliminary minimum estimate.

5. Repeat Steps 1-4 for each biological indicator.

The AET approach for a group of field-collected sediment samples is shown in Figure 10-1. The samples were collected at various locations and were analyzed for 1) toxicity in a laboratory bioassay and 2) the concentrations of a suite of chemicals, including lead and 4-methylphenol. Based on the results of bioassays conducted on the sediments from each station, two subpopulations of all sediments are represented by bars in the figure:

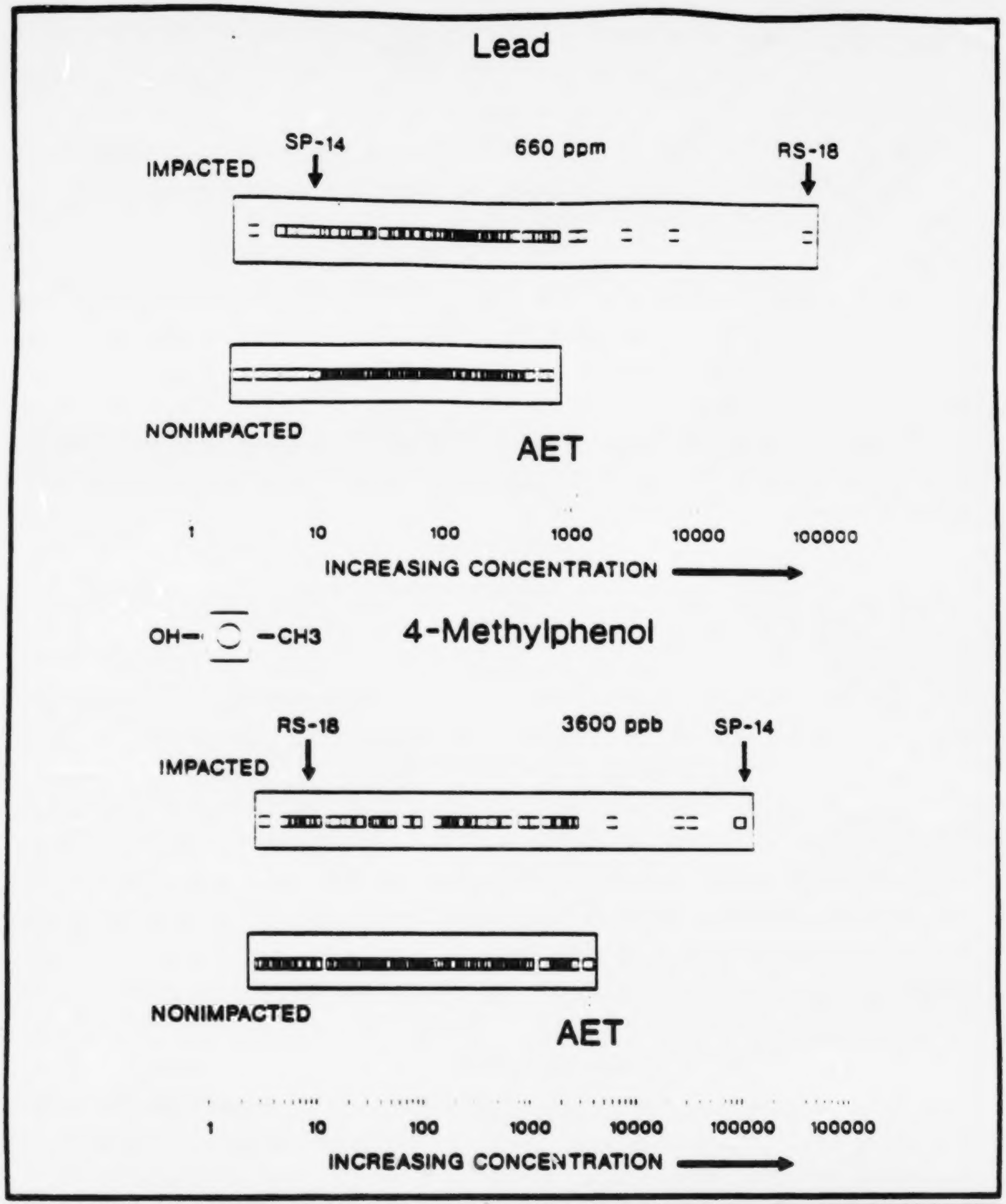

Lead
SP-14
660 ppm
RS-18
IMPACTED
NONIMPACTED
AET
1
10
100
1000
10000
100000
INCREASING CONCENTRATION
OH—
—CH3
4-Methylphenol
RS-18
3600 ppb
SP-14
IMPACTED
NONIMPACTED
AET
1
10
100
1000
10000
100000
100000
INCREASING CONCENTRATION

Figure 10-1. The AET approach applied to sediments tested for lead and 4-methylphenol concentrations and toxicity response during bioassays.

- Sediments that did not exhibit statistically significant ($P>0.05$) toxicity relative to reference conditions ("non-impacted" stations)

- Sediments that exhibited statistically significant ($P\leq0.05$) toxicity in bioassays relative to reference conditions ("impacted" stations).

Over the observed range of concentrations for these sediment samples (horizontal axis in Figure 10-1), the sediments fall into two groups for each chemical:

- At low to moderate concentrations, significant sediment toxicity occurred in some samples, but not in others

- At concentrations above an apparent threshold value, significant sediment toxicity occurred in all samples.

The AET value is defined for each chemical as the highest concentration of that chemical in the sediments that did not exhibit sediment toxicity. Above this AET value, significant sediment toxicity was always observed in the data set examined. Data are treated in this manner to reduce the weight given to samples in which factors other than the contaminant examined (e.g., other contaminants, environmental variables) may be responsible for the biological effect.

For each chemical, additional AET values could be defined for other biological indicators that were tested (e.g., other bioassay responses or depressions in the abundances of certain indigenous benthic infauna).

2.1.1 Objectives and Assumptions--

The objective of the AET approach is to identify concentrations of contaminants that are associated exclusively with sediments exhibiting statistically significant biological effects relative to reference sediments. AET value generation is a conceptually simple process and incorporates the complexity of biological-chemical interrelationships in the environment without relying upon *a priori* assumptions about the mechanisms of these interrelationships. Although the AET approach does not require specific assumptions about mechanisms of the uptake and toxic action of chemicals, it does rely on more general assumptions regarding the interpretation of matched biological and chemical data for field-collected samples, as described below:

- For a given chemical, concentrations can be as high as the AET value and not be associated with statistically significant biological effects (for the indicator on which the AET was based)

- When biological impacts are observed at concentrations below an AET value for a given chemical, it is assumed that the impacts may be related to another chemical, chemical interactive effects, or other environmental factors (e.g., sediment anoxia)

- The AET concept is consistent with a relationship between increasing concentrations of toxic chemicals and increasing biological effects (as observed in laboratory exposure studies).

The assumptions in interpreting environmental data are demonstrated below with actual field data. Using Figure 10-1 as an example, sediment from Station SP-14 exhibited severe toxicity, potentially related to a greatly elevated concentrations of 4-methylphenol (7,400 times reference levels). The same sediment from Station SP-14 contained a relatively low concentration of lead that was well below the AET for lead (Figure 10-1). Despite the toxic effects associated with the sample, sediments from many other stations with higher lead concentrations than Station SP-14 exhibited no statistically significant biological effects. These results were interpreted to suggest that the effects at Station SP-14 were potentially associated with 4-methylphenol (or a substance with a similar environmental distribution) but were less likely to be associated with lead. A converse argument can be made for lead and 4-methylphenol in sediments from Station RS-18.

Applied in this manner, the AET approach helps to identify measured chemicals that are potentially associated with observed effects at each biologically impacted site and eliminates from consideration chemicals that are far less likely to be associated with effects (i.e., the latter chemicals have been observed at higher concentrations at other sites without associated biological effects). Based on the results for lead and 4-methylphenol, bioassay toxicity at five of the impacted sites shown in the figure may be associated with elevated concentrations of 4-methylphenol, and toxicity at eight other sites may be associated with elevated concentrations of lead (or similarly distributed contaminants).

As illustrated by these results, the occurrence of biologically impacted stations at concentrations below the AET of a single chemical does not imply that AET values in general are not protective against biological effects, only that single chemicals may not account for all stations with biological effects. By developing AETs for multiple chemicals, a high percentage of all stations with biological effects are accounted for with the AET approach (see Section 3.2.9 and U.S. EPA 1988).

AETs can be expected to be more predictive when developed from a large, diverse database with wide ranges of chemical concentrations and a wide diversity of measured chemicals. Data sets that have large concentration gaps between stations and/or do not cover a wide range of concentrations must be scrutinized carefully (e.g., to discern whether chemical concentrations in the data set exceed reference concentrations) to determine whether AET generation is appropriate.

2.1.2. Level of Effort--

2.1.2.1 Type of Sampling Required--Collection of field data for initial generation of AET is a labor-intensive and capital-intensive process. The exact level of sampling effort required depends on the amount and variety of data collected (e.g., the number of samples collected, the diversity of biological indicators that are tested, and the range of chemicals measured). One means of minimizing these costs is to compile existing data that meet appropriate quality assurance criteria. There are no definitive requirements for the size and variety of the database, although a study of the predictive abilities of the AET approach with Puget Sound data (Barrick et al. 1988) resulted in the following recommendations for data collection:

- Collect or compile chemical and biological effects data from 50 stations or more (and from suitable reference areas).

- Bias the positioning of stations to ensure sampling of various contaminant sources (e.g., urban environments with a range of contaminant sources and, preferably, with broad geographic distribution) over a range of contaminant concentrations (preferably over at least 1-2 orders of magnitude).

- Conduct chemical tests for a wide range of chemical classes (e.g., metals, nonionic organic compounds, ionizable organic compounds). To generate AETs on an organic carbon-normalized basis, total organic carbon (TOC) measurements are required in all sediments.

- Ensure that detection limits of <100 ppb (lower if possible) are attained for organic compounds. High detection limits (i.e., insensitive analyses) can obscure the occurrence of chemicals at low to moderate concentrations; as noted previously, only detected data are used in AET calculations. Metals are naturally occurring substances and most metals concentrations typically exceed routine detection limits.

The only strict requirement for field sampling of data for AET generation is the collection of "matched" chemical and biological data (as described at the beginning of Section 2.1). Matched data sets should be used to reduce the possibility that uneven (spatially variable) sediment contamination could result in associating biological and chemical data that are based on dissimilar sediment samples. Because the toxic responses of stationary organisms (e.g., bioassay organisms confined to a test sediment, or infaunal organisms largely confined to a small area) are assumed to be affected by direct association with contaminants in the surrounding environment, it is considered essential that chemical and biological data be collected from nearly identical subsamples from a given station.

2.1.2.2 Methods--Methodological details for the generation of AET values are described at the beginning of Section 2.1.

2.1.2.3 Types of Data Required--There are two fundamental kinds of data analysis required for AET generation:

- Statistical analysis of the significance of biological effects relative to reference conditions (i.e., classification of stations as impacted or nonimpacted for each biological indicator)

- Generation of an AET value for each chemical and biological indicator (essentially a process of ranking stations based on chemical concentration).

Additional kinds of data analysis needed for AET generation are quality assurance/quality control (QA/QC) review of biological and chemical data, and evaluation of the appropriateness of reference area stations. These topics have been described elsewhere (e.g., Beller et al. 1986; Barrick et al. 1988).

The AET method does not intrinsically require a specific method of statistical analysis for determination of significance of biological effects relative to reference conditions. Existing Puget Sound AET have relied largely on pairwise t-tests; details of statistical analyses performed for the generation of Puget Sound AET have been described elsewhere (U.S. EPA 1988; Barrick et al. 1988; Beller et al. 1986). For example, the following steps were used to determine the statistical significance of amphipod mortality bioassay results (Swartz et al. 1985) in field-collected sediments:

- All replicates from all stations in the reference area used for each study were pooled, and a mean bioassay response and standard deviation were calculated

- Results from each potentially impacted site were then compared statistically with the reference conditions using pairwise analysis

- The F_{max} test (Sokal and Rohlf 1969) was used to test for homogeneity of variances between each pair of mean values

- If variances were homogenous, then a t-test was used to compare the two means

- If variances were not homogenous, then an approximate t-test (Sokal and Rohlf 1969) was used to compare the two means

- Statistical significance was tested with a pairwise error rate of 0.05 to ensure consistency among studies of differing sample sizes.

Data analyses that have been applied to other biological indicators are described elsewhere (Beller et al. 1986; Barrick et al. 1988). Notably, comparisons to reference conditions were somewhat more complicated for benthic infaunal abundances than for sediment bioassays. For benthic infaunal comparisons, reference data for each potentially impacted site were categorized so that comparisons were made with samples collected during the same season, at a similar depth, and whenever possible, in sediments with similar particle size characteristics (i.e., percentage of particles <64 um) as those of the potentially impacted site. In this manner, statistical comparisons were normalized to account for the influence of three of the major natural variables known to influence the abundance and distribution of benthic macroinvertebrates. All benthic data were also log-transformed so that data distributions conformed to the assumptions of the parametric statistical tests that were applied. Additional data treatment methods presented elsewhere (Barrick et al. 1988) are not discussed further herein, because they are not considered intrinsic to the AET approach, but rather are options to address potentially unusual matrices or biological conditions.

2.1.2.4 Necessary Hardware and Skills--The primary skills required for AET generation are related to the development of the biological/chemical database. Expertise in environmental chemistry is required to evaluate chemical data quality, and the need for normalization of chemical data and related factors. Biological and statistical expertise are required for the determination of statistical significance. For benthic data in particular, evaluation of appropriate reference conditions and knowledge of benthic taxonomy and ecology are necessary.

Computers are recommended for the efficient generation of AET values. A menu-driven database (SEDQUAL) has been developed for U.S. EPA Region X that is capable of a number of data manipulation tasks, including the following: 1) storing chemical and biological data, 2) calculating AET values, 3) comparing a specified set of AET to stored sediment chemistry data to identify stations at which adverse biological effects are or are not predicted, and 4) based on such comparisons, calculating the rate of correct prediction of biological impacts. The SEDQUAL system, which requires an IBM-AT compatible computer with a hard disk, has been documented in detail in a users manual (Nielsen 1988). The SEDQUAL database currently includes stored data from Puget Sound (over 1,000 samples, not all of which have biological and chemical data) and is available at no cost from U.S. EPA Region X.

2.1.3 Adequacy of Documentation--

Various aspects of the AET approach have been extensively documented in reports prepared for U.S. EPA and other regulatory agencies, as listed below and in the reference list:

- Generation of Puget Sound AET values and evaluation of their predictive ability (Beller et al. 1986; Barrick et al. 1988)

- Data used to generate Puget Sound AET values (appendices of Beller et al. 1986 and field surveys cited in Beller et al. 1986 and Barrick et al. 1988)

- Briefing report to the U.S. EPA Science Advisory Board (U.S. EPA 1988)

- Policy implications of effects-based marine sediment criteria (PTI 1987).

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

The AET approach has been designed for use in evaluating potential adverse impacts to aquatic life associated with chemical contamination of sediments. By empirically determining the association between chemical contamination and adverse biological effects, predictions can be made regarding the levels of contamination that are always associated with adverse effects (i.e., the AET values). These critical levels of contamination can then be used to develop guidelines for protecting aquatic life (e.g., sediment quality values). AETs can be developed for any kind of aquatic organism for which biological responses to chemical toxicity can be measured. The protectiveness of the AET can therefore be ensured by evaluating organisms and biological responses with different degrees of sensitivity to chemical toxicity. For example, evaluations of metabolic changes (i.e., usually a very sensitive biological response) in a pollution-sensitive species would likely result in AET values that are lower and more protective than evaluations of mortality (i.e., generally a less sensitive response) in a more pollution-tolerant species. The protectiveness of AET can also be ensured through the application of "safety factors." For example, to be protective of chronic biological responses, a factor based on an acute-chronic ratio could be applied to AET developed on the basis of acute biological responses.

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

The AET approach is not intrinsically limited in application to specific chemicals or chemical groups. In general, the approach can be used for chemicals for which data are available. However, when using a specific data set to generate AET, it is preferable that AET generation be limited to chemicals with wide concentration ranges (e.g., ranging from reference concentrations to concentrations near direct sources) and/or with appropriate detection frequencies (e.g., greater than 10 detections). A partial list of chemicals for which AET have been developed is presented in Table 10-1.

3.0 USEFULNESS

3.1 Environmental Applicability

3.1.1 Suitability for Different Sediment Types--

The AET approach can be applied to any sediment type in saltwater or freshwater environments for which biological tests can be conducted. By normalizing chemical concentrations to appropriate sediment variables (e.g., percent organic carbon), differences between different sediment types can be minimized in the generation of AET. In practice, identification of unique or atypical sediment matrices is important in determining the general applicability of AET values generated from a specific set of data.

Differences in physical characteristics (e.g., grain size, habitat exposure) is one major factor that may account for stations that do not meet predictions based on existing AET values. In Puget Sound studies, for example, fine-grained sediments dominated stations that had significant amphipod mortality but were not predicted to be so, and coarse-grained sediments dominated stations that had significant depressions in benthic infauna but were not predicted as impacted by benthic AET (Barrick et al. 1988).

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

There are no constraints on the types of chemicals for which AET can be developed. An AET can be developed for any measured chemical (organic or inorganic) that spans a wide concentration range in the data set used to generate AETs. The availability of a wide diversity of chemical data increases the probability that toxic agents (or chemicals that covary in the

TABLE 10-1. SELECTED CHEMICALS FOR WHICH AET HAVE BEEN DEVELOPED IN PUGET SOUND

Metals

- Antimony
- Arsenic
- Cadmium
- Chromium
- Copper
- Lead
- Mercury
- Nickel
- Silver
- Zinc

Organic Compounds

- Low molecular weight PAH
    - Naphthalene
    - Acenaphthylene
    - Acenaphthene
    - Fluorene
    - Phenanthrene
    - Anthracene
    - 2-Methylnaphthalene
- Chlorinated benzenes
    - 1,3-Dichlorobenzene
    - 1,4-Dichlorobenzene
    - 1,2-Dichlorobenzene
    - 1,2,4-Trichlorobenzene
    - Hexachlorobenzene (HCB)
- Total PCBs
- Pesticides
    - p,p'-DDE
    - p,p'-DDD
    - p,p'-DDT
- Miscellaneous Extractables
    - Benzyl alcohol
    - Benzoic acid
    - Dibenzofuran
    - Hexachlorobutadiene
    - N-Nitrosodiphenylamine

High molecular weight PAH

- Fluoranthene
- Pyrene
- Benz(a)anthracene
- Chrysene
- Benzofluoranthenes
- Benzo(a)pyrene
- Indeno(1,2,3-c,d)pyrene
- Dibenzo(a,h)anthracene
- Benzo(g,h,i)perylene

Phthalates

- Dimethyl phthalate
- Diethyl phthalate
- Di-n-butyl phthalate
- Butyl benzyl phthalate
- Bis(2ethylhexyl)phthalate
- Di-n-octyl phthalate

Phenols

- Phenol
- 2-Methylphenol
- 4-Methylphenol
- 2,4-Dimethylphenol
- Pentachlorophenol

Volatile Organics

- Tetrachloroethene
- Ethylbenzene
- Total xylenes

environment with toxic agents) can be included in interpreting observed biological impacts.

To date, AET have been developed for over 60 chemicals frequently detected in the environment, including 16 polycyclic aromatic hydrocarbons (PAH); several alkylated PAH and related nitrogen-, sulfur-, and oxygen-containing heterocycles; polychlorinated biphenyls (PCBs) (reported as total PCBs); 5 chlorinated benzenes; 6 phthalate esters; 3 chlorinated hydrocarbon pesticides; phenol and 4 alkyl-substituted and chlorinated phenols; and 10 metals and metalloids; 3 volatile organic compounds; and 5 miscellaneous extractable substances. Data for other miscellaneous chemicals that were less frequently detected or analyzed for in the Puget Sound area were also evaluated for their potential use in developing AETs (e.g., resin acids and chlorinated phenols in selected sediments from areas influenced by pulp and paper mill activity).

AETs have been developed for chemical concentrations normalized to sediment dry weight and sediment organic carbon content (expressed as percent of dry weight sediment). Using a 188-sample data set from Puget Sound, AETs were also developed for data normalized to fine-grained particle content (expressed as the percent of silt and clay, or <63-um particulate material, in dry weight of sediment). These latter AET values did not appear to offer advantages in predictive reliability over the more commonly used dry weight and TOC normalizations (Beller et al. 1986).

3.1.3 Suitability for Predicting Effects on Different Organisms--

The AET approach can be used to predict effects on any life stage of any marine or aquatic organism for which a biological response to chemical toxicity can be determined. Because the approach relies on empirical information that measures the chemical concentrations associated with samples exhibiting adverse effects, the results are directly applicable to predicting effects on the organisms used to generate the AET. The results can also be used to predict effects on nontarget organisms by ensuring that the organisms used to generate an AET are either representative of the nontarget organisms or are more sensitive to chemical toxicity than those organisms. For example, AETs generated for a species of sensitive amphipod might be considered as protective of the chemical concentrations associated with adverse effects in other species of equally or less sensitive amphipods. At the same time, these AET might be considered protective of most other benthic macroinvertebrate taxa, because they are based on a member of a benthic taxon (i.e., Amphipoda) that is considered to be sensitive to chemical toxicity (Bellan-Santini 1980). By contrast, AETs generated for a pollution-tolerant species such as the polychaete *Capitella capitata* (cf. Pearson and Rosenberg 1978), might be considered representative for other pollution-tolerant species, but not protective for most other kinds of benthic macroinvertebrates.

3.1.4 Suitability for In-Place Pollutant Control--

In remedial action programs, assessment tools such as the AET approach can be used to address the following specific regulatory needs:

- Provide a preponderance-of-evidence for narrowing a list of problem chemicals measured at a site
- Provide a predictive tool for cases in which site-specific biological testing results are not available
- Enable designation of problem areas within the site
- Provide a consistent basis on which to evaluate sediment contamination and to separate acceptable from unacceptable conditions

- Provide an environmental basis for triggering sediment remedial action

- Provide a reference point for establishing a cleanup goal.

Because AET values are derived from sediments with multiple contaminants, they incorporate the influence of interactive effects in environmental samples. The ability to incorporate the influences of chemical mixtures, either by design or default, is an advantage for the assessment of in-place pollutants.

3.1.5 Suitability for Source Control--

The AET approach is well suited for identifying problem areas. Because specific cause-effect relationships are not proven for specific chemicals and biological effects, remedial actions should not be designed exclusively for a specific chemical (this caution applies to all approaches because of the complex mixture of contaminants in environmental samples). The link between problem areas and potential sources of contamination is established by analysis of concentration gradients of contaminants in these problem areas and the presence and composition of contaminants in sediments and source materials. The AET approach provides a means of narrowing the list of measured chemicals that should be considered for source control and provides supportive evidence for eliminating chemicals from consideration that appear to be present at a concentration too low to be associated with adverse biological effects. Reduction of the overall contaminant load to a problem area such that all measured chemicals are below their respective AET is predicted to result in mitigation of the adverse biological effects. It is possible that such source controls may be effective because of the concomitant removal of an unmeasured contaminant.

3.1.6 Suitability for Disposal Applications--

The evaluation of potential biological impacts associated with the disposal of dredged material is an important component in the designation of disposal sites and review of disposal permits for dredged material. AET values provide a preponderance-of-evidence in determining a "reason to believe" that sediment contamination could result in adverse biological effects. Hence, the AET approach is a useful tool for assessing the need for biological testing during the evaluation of disposal alternatives. It is assumed that AET values generated for in-place sediments provide a useful prediction of whether adverse biological effects will or will not occur in dredged material after disposal at aquatic sites.

3.2 General Advantages and Limitations

3.2.1 Ease of Use--

In this section, "use" is treated as both generation and application. The ease of generating AET values depends on the status of the data to be used for AET generation (i.e., whether field data have been collected and whether statistical significance has been determined for biological indicators). It is recommended that a search for existing data be conducted as part of determining the need for collecting new samples. The existing database of matched biological and chemical data from Puget Sound comprises over 300 samples. Collection of new field data (e.g., for application outside of Puget Sound) would require a considerable expenditure of effort, as would the statistical analysis of a large number of samples. However, if data are available and statistical analyses have been performed, the generation of AET values is very easy with the SEDQUAL database (described in Section 2.1.2.4). The menu-driven system allows for a considerable amount of flexibility in choosing stations and biological indicators to be included in AET generation. Application of AET (i.e., comparison of AET values to chemical concentrations in field samples) is also very easy when using SEDQUAL, provided that the field data have been computerized. Application of AET values to chemical data presented in existing literature is also straightforward.

3.2.2 Relative Cost--

The cost of developing AET values can span a wide range, depending on the stage of database development and the numbers and kinds of chemicals and biological indicators used. The least costly means of developing the values is to use existing chemical and biological information, thus minimizing the expenses associated with field sampling and laboratory analyses. (Selective sampling to confirm if existing AET values are applicable would still be useful.) The historical database could be based on the pooled results from various studies conducted in a region, providing that each study passed QA/QC performance criteria and satisfied the prerequisites of the AET approach (e.g., matched chemical and biological measurements and the ability to discriminate adverse biological effects).

If the historical database is judged inadequate to generate AET for a region, then the costs of field measurements of chemical concentrations in sediments and associated biological effects must be incurred to develop the database. These costs can vary substantially, depending on the chemicals and biological indicators evaluated. Costs would be minimized if evaluations were based on a limited range of chemicals and a single, inexpensive biological test. It is recommended that the approach be based on a relatively wide range of chemicals, and if possible, several kinds of biological indicators.

The existing database for the Puget Sound region is based on a wide range of chemicals (i.e., U.S. EPA priority pollutants and other selected chemicals) and four kinds of biological indicators. The costs for developing AET varied considerably among the four indicators. For example, laboratory costs for the least expensive indicator (i.e., Microtox bioassay) were approximately $200 per station, whereas costs for the most expensive indicator (i.e., abundances of benthic macroinvertebrates) were as high as $1,800 per station. Therefore, within the existing database, the range of costs for biological testing spanned almost 1 order of magnitude.

Once AET values have been generated, use of these values to predict the occurrence of biological effects is relatively inexpensive. Chemical data may be compared to AET values by using the SEDQUAL database or through manual data manipulations.

3.2.3 Tendency to be Conservative--

The empirical, field-based nature of the AET approach precludes definitive *a priori* predictions of its tendency to be either over- or underprotective of the environment. The occurrence of biologically impacted stations at concentrations below the AET of a given chemical (see Figure 10-1) may appear to be underprotective. However, the occurrence of impacted stations at concentrations below the AET of a single chemical does not imply that AETs in general are not protective against biological effects, only that single chemicals may not account for all stations with biological effects. If AETs are developed for multiple chemicals, the approach can account for a high percentage of stations with adverse biological effects.

To date, AETs have been developed for acute sediment bioassays of mortality in adult amphipods, developmental abnormality in larval bivalves, and metabolic alterations in bacteria. All of these organism/endpoint combinations are considered to be sensitive to chemical toxicity. AETs have also been generated for *in situ* reductions in the abundances of benthic macroinvertebrates. Because these reductions incorporate chronic (i.e., long-term) exposure to contaminants, they can also be considered as sensitive measures of the effects of chemical toxicity. However, a more protective approach would be to use the lowest of the four kinds of AET for each chemical as the concentration upon which predictions are made. Alternatively, the protectiveness of any kind of AET could be modified by developing sediment quality values based on "safety factors" applied to existing AETs.

3.2.4 Level of Acceptance--

The AET approach has been accepted by several federal and state agencies in the Puget Sound region as one tool in providing guidelines for regulatory decisions. U.S. EPA has used AET values to develop sediment quality values with which to evaluate the potential toxicity of contaminated sediments in urban bays. PSDDA has used AET values as a tool to develop chemical guidelines for determining whether biological testing is necessary for dredged sediments proposed for unconfined, open-water disposal. Ecology has used AET to guide several stages of remedial action and to draft sediment standards for classifying sediments according to their potential for causing adverse biological effects. In several of these applications, AET have been modified by "safety factors" to enhance their protectiveness.

Several major characteristics influence the acceptability of the AET approach. The most attractive characteristic of the approach is probably the reliance on empirical information based on field-collected sediments or indigenous organisms, and exposure of laboratory test organisms to environmental samples. A second attractive feature of the approach is the setting of an AET at the chemical concentration in the data set above which adverse biological effects are always observed. This characteristic provides consistency that, with a representative database used to generate AETs, enhances the preponderance-of-evidence of adverse effects in the environment. The AET values can be updated as new information is collected. The AET approach can also be applied to an existing database in new regions, providing certain prerequisites are met by the database (e.g., synoptic measurement of chemical and biological data, and QA/QC guidelines).

A limitation of the AET approach is that field-based approaches do not directly assess cause-effect relationships. Because sediments in the environment are often contaminated with a complex mixture of chemicals, it is difficult when using field-collected sediment for any approach to relate observed biological effects to a single chemical. The approach also requires selection of appropriate normalized chemical data to address the bioavailability of contaminants to organisms. Organic carbon-normalization may be most appropriate for nonpolar organic contaminants based on theoretical considerations. In addition, nonprotective AETs could be generated if unusual matrices (e.g., slag) that anomalously restrict bioavailability are included in the database used to generate the AETs, or if biological test results are incorrectly classified. Recommended data treatment guidelines for chemical and biological data are discussed by Barrick et al. (1988). The AET approach is currently under review by the U.S. EPA Science Advisory Board.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

If applicable data do not already exist, the development of AET values requires a relatively extensive amount of field sampling and laboratory analysis. The chemical analyses required for development of AET represent standard analytical procedures. A laboratory with appropriately trained staff should be able to conduct the necessary benthic community analyses and sediment bioassays. Specific methods for performing the chemical and biological tests that were used to develop Puget Sound AET are detailed in the Puget Sound Protocols (Tetra Tech 1986). These efforts can be minimized by using historical data whenever possible. Once AETs are developed, their routine implementation is relatively easy. In addition, they can be easily updated as additional data become available.

3.2.6 Level of Effort Required to Generate Results--

As noted in Section 3.2.1, the SEDQUAL database facilitates AET generation and application. After field data have been collected, the most time-consuming task is data entry and verification. Entry of chemical and biological data for 50 samples requires roughly 16 person-hours (assuming 75 chemicals have been measured and biological effects are being coded simply as "impacted" or "nonimpacted"). Generating a set of AET values for a given biological indicator, 75 chemicals, and 50 stations takes approximately 0.75-1 h of computer time on SEDQUAL (and about 5 min of labor to set up the analysis). To compare a set of AET (for 75 chemicals) to a 50-sample set of field data takes approximately 0.5-0.75 h of computer time on SEDQUAL (and roughly 5 min of labor to set up the analysis). SEDQUAL is capable of comparing any kind of chemical sediment criteria to field data, but requires that the numerical criteria be entered in the database.

3.2.7 Degree to Which Results Lend Themselves to Interpretation--

The manner in which the AET approach can be used to interpret matched biological and chemical data from field-collected sediments is described in Section 2.1. As noted previously, the use of AET can help investigators eliminate chemicals from further consideration (as the cause of an observed effect); however, the approach cannot identify specific cause-effect relationships. Because the AET approach is empirical, it is not well-suited to identifying specific toxic agents or elucidating mechanisms of biological uptake and metabolism. However, certain general relationships could be examined on an *a posteriori* basis with the AET approach (e.g., testing the relative importance of different ways of normalizing chemical concentration data in predicting adverse biological effects).

A number of environmental factors may complicate the interpretation of the data. Although the AET concept is simple, the generation of AET values based on environmental data incorporates many complex biological-chemical interrelationships. For example, the AET approach incorporates the net effects of the following factors that may be important in field-collected sediments:

- Interactive effects of chemicals (e.g., synergism, antagonism, and additivity)

- Unmeasured chemicals and other unmeasured, potentially adverse variables

- Matrix effects and bioavailability (i.e., phase associations between contaminants and sediments that affect bioavailability of the contaminants, such as the incorporation of PAH in soot particles).

The AET approach cannot quantify the individual contributions of interactive effects, unmeasured chemicals, or matrix effects in environmental samples, but AET values may be influenced by these factors. AET values are expected to be reliable predictors of adverse effects that could result from the influence of these environmental factors, if the samples used to generate AETs are representative of samples for which AET predictions are made. Alternatively, isolated occurrences of such environmental factors in a data set used to generate AETs may limit the predictive reliability of those AET values. If confounding environmental factors render the AET approach unreliable, then this should be evident from validation tests in which biological effects are predicted in actual environmental samples.

A more detailed discussion of the interpretation of AETs and the confounding effects of environmental factors is presented in U.S. EPA (1988).

3.2.8 Degree of Environmental Applicability--

The AET approach has a high degree of environmental applicability based on its reliance on chemical and biological measurements made directly on environmental samples. Such information provides tangible evidence that various chemical concentrations either are or are not associated with adverse biological effects in typically complex environmental settings.

The environmental applicability of the AET approach has been quantified for the four kinds of AET developed for Puget Sound by evaluating the reliability with which each kind of AET predicted the presence or absence of adverse biological effects in field samples collected from Puget Sound (U.S. EPA 1988). The overall reliability of the four tests ranged from 85 to 96 percent, indicating that all four kinds of AET were relatively accurate at predicting the presence or absence of effects for samples from the existing database. This high level of reliability suggests that AET have a relatively high degree of environmental applicability in Puget Sound, and has been a primary factor for the use of the AET approach by agencies in the Puget Sound region. AET values generated for Puget Sound have also been used as examples of effects-based sediment criteria to provide an initial estimate of the magnitude of potential problem areas in coastal regions of the U.S. for the U.S. EPA Office of Policy Analysis (PTI 1987).

3.2.9 Degree of Accuracy and Precision--

In this section, accuracy is considered to be the ability of AET to predict biological effects and precision represents the expected variability (uncertainty range) for a given AET value for a given data set.

In previous evaluations of the AET approach and other sediment quality values using field-collected data, the accuracy of the approach was defined by two qualities:

- Sensitivity in detecting environmental problems (i.e., are all biologically impacted sediments identified by the predictions of the chemical sediment criteria?)

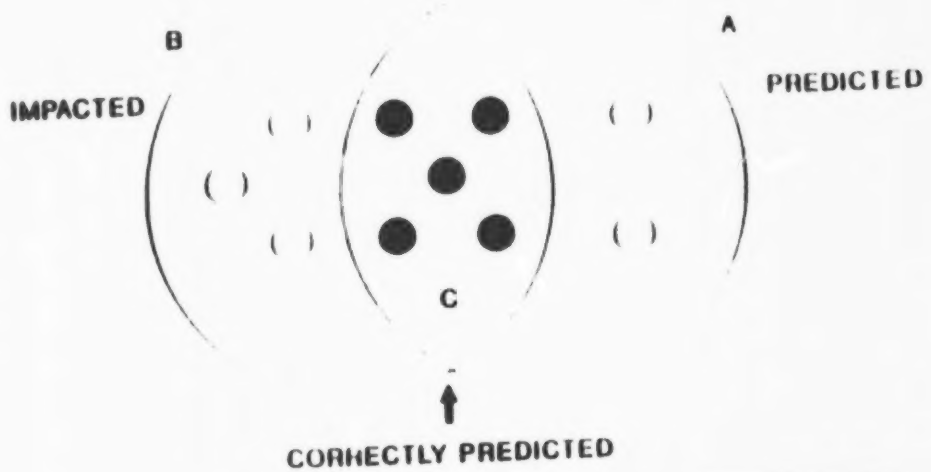
- Efficiency in screening environmental problems (i.e., are only biologically impacted sediments identified by the predictions of the chemical sediment criteria?).

Sensitivity is defined as the proportion of all stations exhibiting adverse biological effects that are correctly predicted using sediment criteria. Efficiency is defined as the proportion of all stations predicted to have adverse biological effects that actually are impacted. Ideally, a sediment criteria approach should be efficient as well as sensitive. For example, a sediment criteria approach that sets values for a wide range of chemicals near their analytical detection limits will likely be conservative (i.e., sensitive) but inefficient. That is, it will predict a large percentage of sediments with biological effects. It will also predict impacts at many stations where there are no biological effects, but chemical concentrations are slightly elevated. The concepts of sensitivity and efficiency are illustrated in Figure 10-2.

The overall reliability of any sediment criteria approach addresses both sensitivity and efficiency. This measure is defined as the proportion of all stations for which correct predictions were made for either the presence or absence of adverse biological effects:

$$\text{Overall reliability} = \frac{\begin{array}{c}\text{All stations correctly predicted as impacted}\\ + \\ \text{All stations correctly predicted as nonimpacted}\end{array}}{\text{Total number of stations evaluated}}$$

High reliability results from correct prediction of a large percentage of the impacted stations (i.e., high sensitivity, few false negatives) and correct


B
A
IMPACTED
PREDICTED
C
CORRECTLY PREDICTED

SENSITIVITY = C/B x 100 = 5/8 x 100 = 63%

EFFICIENCY = C/A x 100 = 5/7 x 100 = 71%

FOR A GIVEN BIOLOGICAL INDICATOR

- A ALL STATIONS PREDICTED TO BE IMPACTED
- B ALL STATIONS KNOWN TO BE IMPACTED
- C ALL STATIONS CORRECTLY PREDICTED TO BE IMPACTED

Figure 10-2. Measures of reliability (sensitivity and efficiency).

prediction of a large percentage of the nonimpacted stations (i.e., high efficiency, few false positives).

An assessment of AET reliability was recently conducted using a large database comprising samples from 13 Puget Sound embayments (Barrick et al. 1988). These evaluations suggest that the AET approach is relatively sensitive for the biological indicators tested and also relatively efficient. For example, 68-83 percent sensitivity and 55-75 percent efficiency were observed when AET generated from a 188-sample data set were evaluated with an independent 146-sample data set. The ranges of sensitivity and efficiency cited above represent the ability of benthic infaunal AET values to predict statistically significant depressions in the abundances of benthic infauna in field-collected samples and the ability of amphipod mortality bioassay AET values to predict statistically significant mortality in bioassays conducted on field-collected sediment.

Precision of the AET approach has not been as intensively investigated as accuracy. AET values are the result of parametric statistical procedures (i.e., determination of the significance of biological effects relative to reference conditions) and nonparametric methods (e.g., ranking of stations by concentration), and thus are not amenable to the routine definition of confidence intervals. However, the degree of AET precision is considered to depend on the following factors:

- The concentration range between the AET (determined by a nonimpacted station) and the next highest concentration that is associated with a statistically significant effect

- Classification error associated with the statistical significance of biological indicator results (i.e., whether a station is properly classified as impacted or nonimpacted, as related to Type I and Type II statistical error)

- The weight of evidence or number of observations supporting a given AET value
- The analytical error associated with quantification of chemical results.

Detailed discussion of these factors is provided in Beller et al. (1986).

One approach used in Puget Sound to estimate the uncertainty range around the AET value was to define the lower limit as the concentration at the nonimpacted station immediately below the AET and to define the upper limit as the concentration at the impacted station immediately above the AET. These limits are based largely on probabilities of statistical classification error. For data sets with large concentration gaps between stations, such uncertainty ranges will be wider and precision will be poorer than for data sets with more continuous distributions. The number of stations used to establish an AET would be expected to have a marked effect on AET uncertainty because small data sets would tend to have less continuous distributions of chemical concentrations than large data sets. Based on analyses conducted with Puget Sound data, the magnitude of the AET uncertainty for 10 chemicals or chemical groups that are commonly detected is typically less than one-third to one-half of the value of the AET itself (considering both amphipod mortality bioassay and benthic infaunal AET data). Based on quality assurance information for these data, analytical error is probably a minor component of overall precision, particularly for metals.

4.0 STATUS

4.1 Extent of Use

The AET approach has been used by several agencies in the Puget Sound region to provide guidelines for regulatory decisions. The U.S. EPA has used AET to develop sediment quality values with which to evaluate the potential toxicity of contaminated sediments in urban bays. PSDDA has used the AET approach as a tool for developing guidelines to determine whether biological testing is necessary for dredged sediments proposed for unconfined, open-water disposal. Ecology has used AET values to establish draft sediment standards for classifying sediments according to their potential to cause adverse biological effects. Ecology and U.S. EPA have also used AET values to identify problem chemicals, link contaminated sediments to potential sources, and provide reference points for the establishment of sediment cleanup goals in the Commencement Bay RI/FS.

Several strategies have been developed for using the AET approach for different regulatory purposes in Puget Sound. In the Superfund program locally, the lowest AET (termed LAET) for the four kinds of AETs used in Puget Sound have been used to establish goals for sediment remedial actions. In dredged material assessment, sediment quality values have been developed for use as protective screening chemical levels by applying "safety factors" to the AET. Because biological effects are rarely expected to occur when chemical concentrations are below these screening levels, additional testing of sediments usually is not required. The AET approach also been used to develop maximum chemical levels, above which adverse effects are predicted for all of the biological tests used to generate AETs in Puget Sound. These maximum levels have been set by the highest AET (termed HAET) for the four biological indicators evaluated in Puget Sound.

Outside the Puget Sound region, chemical and biological data from San Francisco Bay, San Diego Bay, and the Southern California Bight are currently being evaluated for use in developing region-specific AETs for the California State Water Resources Control Board. These California AETs will be compared with Puget Sound AET to evaluate similarities and differences between the two kinds of information.

4.2 Extent to Which Approach Has Been Field-Validated

As described in U.S. EPA (1988), the reliability of AETs generated from Puget Sound data was evaluated with tests of sensitivity and efficiency (defined in Section 3.2.9). Tests of the sensitivity and efficiency of the AET approach were carried out in several steps, as described below:

- The chemical database was subdivided into groups of stations that were tested for the same biological effects indicators. Specifically, all chemistry stations with associated amphipod bioassay data were grouped together (287 stations), all chemistry stations with associated benthic infaunal data were grouped together (201 stations), all chemistry stations with associated oyster larvae bioassay data were grouped together (56 stations), and all chemistry stations with associated Microtox bioassay data were grouped together (50 stations). Stations with more than one biological indicator were included in each appropriate group.

- The stations in each group were classified as impacted or nonimpacted based on the appropriate statistical criteria (i.e., F_{max} and t-tests at alpha = 0.05).

- Several tests of reliability were conducted at this point:

  - Test 1: AET values (dry weight) were generated with the entire Puget Sound database available in 1988, and sensitivity and efficiency tests were performed against the same database for each biological indicator.

- Test 2: The test described above was repeated in two parts: (a) using TOC-normalized AET values for nonionic organic compounds and dry weight-normalized AET values for all other compounds (i.e., ionizable organic compounds, metals, and metalloids), and (b) using TOC-normalized data for all chemicals. Test 2 allowed for *a posteriori* evaluation of the relative success of dry weight and TOC normalization for nonionic organic chemicals.

- Test 3: Because the efficiency of the AET based on the entire Puget Sound database is 100 percent by constraint (as in Tests 1 and 2), predictive efficiency was estimated by the following procedure. For each biological indicator, a single station was sequentially deleted from the total database, AETs were recalculated for the remaining data set, and biological effects were predicted for the single deleted station. The predictive efficiency was the cumulative result for the sequential deletions of single stations. For example, the 287-sample database for amphipod bioassay results can be used to provide a 286-sample independent database for predicting (in sequence) effects on all 287 samples.

- Test 4: In this test, independent data sets were used to generate and test AETs to confirm the sensitivity and efficiency measurements in Tests 1 and 3. AETs (dry weight) generated with 188 stations from diverse geographic regions in Puget Sound were tested with a completely independent set of 146 Puget Sound stations.

In addition, the influence of geographic location and other factors on AET predictive ability were examined (Barrick et al. 1988). Further testing of Puget Sound AET values using matched biological/chemical data from other geographic areas is desirable before recommending direct application of the Puget Sound values in other geographic regions.

4.3 Reasons for Limited Use

The AET approach was developed in the Puget Sound region, and has been used to provide agencies with guidelines for evaluating and managing contaminated sediments. The approach is not yet commonly used outside of Puget Sound. Because the approach is based on empirical data, region-specific values should be evaluated thoroughly by experts before application in other regions. For example, because regional reference areas are used to determine the significance of adverse biological effects in the approach, there may be concern that AET developed for one region may be overprotective or underprotective of other areas.

Development of site-specific AET for other geographic areas may require additional sampling. Because many past studies were not multidisciplinary, measurements were often made only for chemistry or biology rather than for both kinds of information. In such cases, there will be a limited amount of appropriate historical data that can be used to develop AETs. The integration or comparison of AET data sets among different regions can also be restricted because appropriate biological indicators for generating AETs may vary among regions.

4.4 Outlook for Future Use and Amount of Development Yet Needed

The following two approaches to AET development could be particularly beneficial in expanding the use of this approach:

- Use of laboratory cause-effect (spiking) studies to evaluate AET predictions on a chemical-specific basis

- Use of a large set of matched biological/chemical data from different geographic areas to test the predictive ability of AET and to test the "precision" of AET values based on data sets from different areas.

The AET method is currently under technical review by the U.S. EPA Science Advisory Board. Based on this review, additional development of the method may be recommended.

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CHAPTER 11. A SUMMARY OF THE SEDIMENT ASSESSMENT STRATEGY RECOMMENDED BY THE INTERNATIONAL JOINT COMMISSION

Philippe Ross
Illinois Natural History Survey
607 East Peabody Drive
Champaign, IL 61820-6970
(217) 244-5054 or (312) 353-0117

The International Joint Commission (IJC) Sediment Subcommittee has published a document entitled *Procedures for the Assessment of Contaminated Sediment Problems in the Great Lakes* (IJC 1988a). An overview of the IJC (1988a) strategy for assessing contaminated sediments is provided in this chapter. However, because it would be inappropriate to reproduce all, or substantially all, of the document in this chapter, the interested reader is referred to the IJC (1988a) document itself for an explanation of details that are not provided herein.

1.0 SPECIFIC APPLICATIONS

1.1 Current Use

The IJC (1988a) document is intended as guidance for the assessment of contaminated sediments in the Great Lakes. Its first application is in the work plan for sediment investigations at Great Lakes areas of concern (AOCs, as identified by the IJC). Section 118(c)(3) of the Water Quality Act of 1987 calls for U.S. EPA's Great Lakes National Program Office to survey at least five AOCs as part of a 5-yr study and demonstration program called ARCS (Assessment and Remediation of Contaminated Sediments). The strategy recommended by IJC (1988a) will be applied through a series of activities involving physical mapping and characterization, sampling, chemical analyses, toxicity testing, and *in situ* community analysis. The assessment will begin in 1989, with completion scheduled for 1991.

1.2 Potential Use

Other AOCs will eventually be evaluated in the process of developing remedial action plans. It is possible that other Great Lakes harbors, rivers, and estuaries will be added to the list of AOCs, in which case remedial action plans would have to be developed there. In addition, the guidance document could potentially be used to assess suspected sediment contamination outside the Great Lakes basin.

2.0 DESCRIPTION

2.1 Description of Method

2.1.1 Objectives and Assumptions--

In response to the need for a common approach to the assessment of contaminated sediments, the IJC's Sediment Subcommittee has developed a strategy based on protocols that emphasize biological monitoring. The approach is intended for use in comprehensive assessments of areas (e.g., bays, harbors, rivers, other depositional zones) where sediment contamination and the need for remedial action are suspected. While the suggested strategy attempts to minimize the cost and expertise, the assessments are relatively large undertakings appropriate to situations where large-scale remedial actions might be contemplated. In such cases, the cost of conducting accurate assessments would be justified if the subsequent remedial options could cost far more than the assessments. It was not the primary intent of the subcommittee to provide guidance for small-scale decision-making activities, such as sample-by-sample disposal of dredged material from navigation channels. Nevertheless, some of the component methods described could be useful and cost-effective in this regard. The first major assumption, therefore, is that the scope of the study in question is sufficient to warrant a large-scale integrated investigation.

Another fundamental assumption is that the ultimate concern of a problem assessment focuses on whether sediment contaminants are exerting biological stress or are being bioaccumulated. Accepting this assumption, it follows that adequate assessments of sediment quality should involve components of chemistry, toxicity, and infaunal community structure (Chapman and Long 1983), a concept frequently referred to as the Sediment Quality Triad approach (see Chapter 9). The proposed strategy has the following objectives:

- To provide accurate assessments of specific problems by using a modified 'triad' approach, which integrates chemical, physical, and biological information
- To perform tasks in a sequence so that the results from each technique can be used to reduce subsequent sampling requirements and costs
- To provide adequate proof of linkage between the contamination and the observed biological impact
- To quantify problem severity, thereby enabling intercomparisons between and within areas of investigation (thus allowing a priority list for remedial actions to be developed and the objective selection of appropriate remedial options)

- To consider the effects on different species and different trophic levels, since biological impairment may occur in the water column and the sediments if resuspension occurs, and since there is no such thing as the universal "most-sensitive species" (Cairns 1986).

The IJC approach is an integrated strategy that provides the necessary data to identify sediment-associated contamination as the problem source, specify effects, rank problem severity, and assist in the selection of remedial options. While the assessment portion of the document identifies a set of the best currently available assessment tools (see Section 2.1.2.2), it is assumed that decisions will be made based on the circumstances unique to each AOC. There is no substitute for experience (expert judgment), and it is also assumed that appropriate expertise will be assembled before the assessment study plan is formulated.

2.1.2 Level of Effort--

2.1.2.1 Type of Sampling Required--The IJC (1988a) approach involves two stages. Stage I, the initial assessment, is used for areas where an inadequate or outdated database exists. Stage I uses only *in situ* assessment techniques and criteria: a limited physical description of the area (e.g., basin size and shape, bathymetry) and the sediments, bulk chemical analyses, resident benthic community organization (e.g., family level identifications), fish contaminant body burdens (one important species, selected by expert judgment), and external abnormalities on collected specimens. Any one of the following criteria provide sufficient justification for proceeding to Stage II:

- Concentrations of metals above background levels in sediments
- Concentrations of hazardous persistent organic compounds above best available detection levels in sediments

- Concentrations of hazardous persistent organic compounds above detection levels in fish or benthos

- The absence of a healthy benthic community (e.g., absence of clean water organisms such as amphipods or mayflies, presence of a community dominated by oligochaetes, the complete absence of invertebrates)

- Presence of external abnormalities in fish.

These conditions must be supported by evidence that the observed situation is not due to a major sediment perturbation, such as dredging or substrate modification.

Available data may preclude the need for a Stage I assessment. The cost and effort that Stage I entails should be avoided if there is already strong evidence of a contamination problem.

When a probable sediment contamination problem is identified, either through the initial assessment or from the examination of existing data, then Stage II, the detailed assessment, should be undertaken. The detailed assessment consists of four phases, which together define the sediment problem in the most cost-effective manner. The phases are not inflexible protocols, but rather logical groupings of work units. The expert investigator should be responsible for the final study design.

In Phase I of Stage II, extensive information on the physical composition of the sediments is collected. These data are used to define areas or zones of homogeneity within a study area. Knowledge of these zones allows sampling requirements for Phase II to be estimated.

In Phase II, the benthic community structure is examined to the lowest possible taxonomic level (e.g., species or variety), along with the surficial sediment chemistry (e.g., pH, total organic carbon, redox potential, metals, extractable organic compounds). Phase II results can be combined with Phase I data to reduce the sampling effort in the next phase.

In Phase III, a battery of laboratory bioassays (e.g., Microtox, algal, daphnid, benthic invertebrate, fish, Ames test) are performed on a smaller number of sediment samples than those in the Phase II sample set. Since fresh sediment must be collected for this phase, precision position-finding equipment is required to relocate previously sampled sites. Phase III costs can be reduced by performing acute lethality bioassays on a sediment sample before proceeding to tests that measure chronic or sublethal effects. Also in Phase III, sediment cores are collected, dated, and sectioned for stratified chemical analyses and bioassays. Finally, adult fish are examined histopathologically for internal (e.g., liver) tumors. In relatively confined geographical areas, Phases II and III may be combined, as further sampling may be more costly than conducting additional bioassays, and relocating Phase II sets for Phase III sampling may be difficult. In this case, Phase II sampling will include extra material for Phase III.

In the fourth and final phase, sediment dynamics (e.g., accumulation, resuspension, movement) and factors affecting them are quantified. All of the foregoing information is necessary for the selection of appropriate remedial options. For example, depositional history, as revealed by sampling sediment cores, and sediment dynamics are critical pieces of information in the selection and cost evaluation of remedial options.

Criteria that clearly indicate when some form of remedial action must be considered (based on the results of Stage II) are essential. Due to the absence of definitive sediment action criteria at time of writing, the criteria proposed by the IJC (1988a) are highly conservative, following the language of the 1978 Great Lakes Water Quality Agreement as revised in 1987 (especially Annexes 1 and 12), in order to promote maximum protection and effective restoration of the Great Lakes ecosystem. The IJC (1988a) urges that these criteria be reviewed regularly to ensure that they continue to fulfill their intended purpose.

2.1.2.2 Methods--During Stage I, the minimum amount of information necessary to assess potential problem sediments is collected. A variety of physical, chemical, and biological measurements are recommended, as outlined below:

- A geographical description of the area and its bathymetry are required.

- Sediment grain size - Size analysis techniques based on settling velocity (American Society for Testing and Materials 1964; Duncan and LaHaie 1979) are recommended. The sand fraction is removed by a 62-um sieve and analyzed separately from the fine-grained material.

- Sediment water content - The water content can be determined during sample preparation for grain size and other analyses by comparison of sample weights before and after either freeze-drying or oven-drying (Adams et al. 1980).

- Redox potential (Eh) and pH should be measured [specific methods are not recommended by IJC (1988a)].

- Organic carbon - It is recommended that total sediment organic carbon be measured as described by Plumb (1981).

- Phosphorus - Two measurements are suggested: total phosphorus as extracted from sediment by sodium carbonate fusion or by perchloric acid digestion, and bioavailable phosphorus as estimated by NaOH extractable phosphorus (Williams et al. 1980).

- Ten metals (lead, nickel, copper, zinc, cadmium, chromium, iron, manganese, mercury, and arsenic) are recommended for routine analysis at Great Lakes AOCs. Additional metal analyses are left to the judgment of the investigator. An extraction procedure using a mix of hydrochloric and nitric acids (1:1) is suggested (Plumb 1981).

- Persistent organic compounds - The reader is referred to the U.S. EPA (1984) protocols for broad scans and analyses of individual compounds. When the strategy was written, no standardized chemical protocols for estimating bioavailability of trace organic compounds were identified.

- External abnormalities in fish - The presence of one or more external abnormalities is often indicative of anthropogenically induced stress or damage. In the case of the brown bullhead, *Ictalurus nebulosus*, phenomena such as stubbed barbels, skin discoloration (melanoma), and skin tumors are highly correlated with liver cancer incidence (Smith et al. 1988). It is recommended that locally occurring catfish (particularly *I. nebulosus*) be examined for tumors, melanoma, blindness, and barbel abnormalities during a Stage I assessment.

- Contaminant body burdens - The benthic infauna are in continuous contact with the sediments, providing a direct measure of the specific relationship between localized sediment contaminant concentrations and bioavailability. Carp are also regularly in contact with and ingest large quantities of sediments. They represent a larger spatial and temporal integration of contaminants than do the benthic infauna. Collection of adult common carp (*Cyprinus carpio*) for tissue residue analysis is recommended. Three to five fish per replicate should be composited. The number of replicates is determined using variability estimates from monitoring programs (Schmitt et al. 1983) and a chosen level of precision, to calculate an idealized sample size (p. 247, Sokal and Rohlf 1969). It is also recommended that the most abundant benthic invertebrate species (often oligochaete worms in contaminated sediments) be sampled in early summer, prior to thermal stratification. Standard U.S. EPA methods are suggested for tissue residue analysis. The problem of obtaining enough biomass for analysis (at least 1 g) is recognized.

- Benthic community structure - In a Stage I assessment, a preliminary analysis of community structure impairment is recommended. A qualitative study with minimal replication and identification only to the family level is suggested. Because it is important that rare taxa be sampled, simple techniques that employ inexpensive equipment but take large samples are recommended. This approach should suffice to identify the existence of a stressed community for the purposes of Stage I criteria (see Section 2.1.2.1 above).

The detailed assessment of Stage II consists of more focused analyses to supplement or complement information obtained in Stage I. Phase I of the detailed assessment focuses on physical mapping of the environment. The most important aspect of the physical assessment of a suspected contaminated sediment deposit is its three-dimensional mapping. A rectangular grid pattern is recommended for the initial mapping operation. Concurrent with bottom sampling at grid intersections, echo-sounder and side-scan sonar surveys should be performed to improve spatial resolution of sediment zones and bottom features. Detailed surveys should include piston coring for stratigraphic resolution. The grid sampling results should be examined using cluster analysis (or similar techniques), which are easy to interpret and functional with a small number of variables. Basic information required in this phase includes geographic location, areal extent, thickness and total sediment volume, average depths of overlying water, and the grain size properties of the deposit. Phase I results are used to select sampling sites for later phases.

Phase II of the detailed assessment focuses on surficial sediment chemistry and benthic community structure. Based on the previous mapping of homogeneous zones (Phase I), effort in Phase II can be expended in depositional areas and in those areas with fine-grained sediments. Surficial chemistry sampling should be coincident with the sampling for detailed benthic community structure analysis. Total organic carbon, redox potential, pH, metals, and persistent organics should be measured. Investigators are referred to Plumb (1981), Williams et al. (1980), and U.S. EPA (1984) for collection and analysis methods.

Since the main objective of Stage II community structure assessment is to examine subtle distinctions in stress response, more detailed taxonomic data are required in this phase than were required in Stage I. In the study design and sample collection steps, investigators are urged to follow the 10 principles of sampling set forth by Green (1979). Further guidance is given in Elliott (1977) for critical factors such as site selection, sample numbers, sampling design, and data analyses. To help investigators assess community impact, IJC (1988a) provides a partial list of literature descriptions of normal nearshore communities in habitats that most closely approximate Great Lakes AOCs. A detailed discussion of statistical methods is also included.

Phase III of the detailed assessment consists of obtaining additional information concerning sediment toxicity (i.e., bioassays and fish histopathology) and stratigraphic characterization of sediment cores. A suite of bioassays is proposed for toxicological evaluation of sediments:

- Microtox - an acute, liquid-phase (elutriate or pore-water) test with luminescent bacteria (Bulich 1984)
- Algal photosynthesis - an acute, liquid-phase test using natural communities [algal fractionation bioassay (Munawar and Munawar 1987)] or the laboratory species *Selenastrum capricornutum* (Ross et al. 1988)
- Zooplankton life-cycle tests (*Daphnia magna* liquid and solid phases) monitoring growth and reproduction (Nebeker et al. 1984; LeBlanc and Surprenant 1985)
- Chronic, solid-phase tests using the benthic invertebrates *Chironomus tentans* (Nebeker et al. 1984), *Hyalella azteca* (Nebeker et al. 1984), or *Hexagenia limbata* (Malueg et al. 1983)
- A solid-phase fish bioaccumulation test with the fathead minnow *Pimephales promelas* (Mac et al. 1984)
- The liquid-phase (extract) Ames *Salmonella*/microsome assay, a bacterial mutagenicity test (Tennant et al. 1987).

In addition to bioassays, histopathological examinations of indigenous adult fish (especially *Ictalurus nebulosus*), focusing on preneoplastic and neoplastic liver lesions (Couch and Harshbarger 1985), are recommended.

Also included in Phase III work are chemical analyses and dating of sediment cores. Isotopic (^{14}C, ^{210}Pb, ^{55}Fe, ^{137}Cs) and biostratigraphic [i.e., ragweed (*Ambrosia*) pollen] methods are both recommended for dating sediment cores. This dating is necessary to establish the three-dimensional configuration of the contaminated sediment mass and to assign a date to the sediment depositional unit.

In Phase IV of the detailed assessment, studies on sediment dynamics are necessary to determine the following:

- Potential water column impacts through resuspension
- Movement of contaminated sediment out of the AOC
- The quality and rate of new sediment accumulation
- Vertical and horizontal redistribution of sediments and their contaminant burdens within an AOC.

This information is essential for the development and evaluation of a remediation plan. In the absence of practical predictive models, suspended sediment characterization (Poulton 1987), shear strength measurements (Terzaghi and Peck 1967), and resuspension studies (Tsai and Lick 1986) are recommended.

2.1.2.3 Types of Data Required--The Stage I initial assessment should be based on aberrant macrozoobenthic community structure (ascertained from family level taxonomic identification); metals concentrations above background levels in the surficial sediments (ascertained from dating);

hazardous persistent organic compound concentrations above detection levels in carp, benthos, or surficial sediments; metals concentrations in carp or benthos, established on a case-by-case basis; and presence in fishes of external abnormalities known to have contaminant-related etiologies.

The Stage II detailed assessment should be based on a phased sampling of the physical, chemical, and biological aspects of the sediments. The biological impacts should be assessed with both field (benthic invertebrate community structure and incidence of fish liver tumors) and laboratory (battery of selected bioassays) methods. The phased sampling approach will allow subsequent testing requirements to be reduced. When Phases I and II have revealed homogeneous zones of sediment type and similar community structure, the number of Phase III samples can be appropriately scaled down. Impairment due to sediment contamination and the probable need for remediation are established when the biomonitoring results from the detailed assessment demonstrate significant departures from controls.

Each section of IJC (1988a) contains a detailed discussion of the statistical procedures required, with references and examples. The preferred method of interpretation is left to the expert investigator in many cases.

2.1.2.4 Necessary Hardware and Skills--The initial assessment, and to an even greater degree the detailed assessment, require a large array of field and laboratory equipment. Although none of the items recommended are unusual or inordinately sophisticated, one laboratory or field unit is unlikely to have all the required apparatus. Specific suggestions for hardware and skills are provided by IJC (1988a). Because this approach is intended for major sediment assessment efforts, several groups would probably have to be mobilized to contribute to the effort.

2.1.3 Adequacy of Documentation--

Each component method described in IJC (1988a) is fully referenced in the text and accompanied by a separate bibliography. Some methods are more developed than others, and areas where additional validation or calibration is needed are clearly identified in the text.

2.2 Applicability of Method to Human Health, Aquatic Life, or Wildlife Protection

The IJC strategy includes direct measures of effects on benthic infauna and fishes, and is thus directly applicable to aquatic biota. Existing sediment assessment methods (e.g., Apparent Effects Threshold, Sediment Quality Triad) could be used to evaluate the results of the Stage II detailed assessment, and to determine whether chemically contaminated sediments have affected aquatic biota in the vicinity of AOCs. Although the IJC (1988a) strategy was not designed to assess the effects of toxic chemicals on wildlife or humans, the tissue residue data and the sediment chemistry data may be useful in preliminary evaluations of contaminant exposure to these populations. Wildlife exposure could occur through consumption of chemically contaminated prey. Human exposure could occur through consumption of chemically contaminated fish or through dermal absorption by direct contact with chemically contaminated sediments or water.

2.3 Ability of Method to Generate Numerical Criteria for Specific Chemicals

The document was designed to provide guidance to assessment programs. Nevertheless, since chemical, toxicological, and infaunal data are collected in the Stage II assessment, it is possible that these data could be used to develop chemical-specific criteria. For example, data from the Stage II assessment could be used to develop empirical sediment quality values (e.g., AET values) that are protective of aquatic biota in locations other than the AOC under consideration.

3.0 USEFULNESS

3.1 Environmental Applicability

3.1.1 Suitability for Different Sediment Types--

The approach recommended in IJC (1988a) is suitable for any sediment type. Indeed, one of its major objectives is to characterize and provide a three-dimensional map of the contaminated sediment mass, including physical, chemical, and biological variables. The investigator is given the flexibility to choose the appropriate sampling methods for the sediment type or types in the AOC under study.

3.1.2 Suitability for Different Chemicals or Classes of Chemicals--

The document is intended for situations where contamination is suspected, but where the toxic chemicals may or may not be identified. The methods recommended by IJC (1988a) are effective for most contaminants found in Great Lakes sediments. The broad-based nature of the approach contains sufficient flexibility to deal with anomalous situations.

3.1.3 Suitability for Predicting Effects on Different Organisms--

The proposed strategy includes both laboratory testing and analysis of indigenous communities (i.e., fish, macrozoobenthos). In this way, laboratory results (i.e., chemistry, toxicity) which can be compared to standard conditions and literature values may be placed in the context of empirically derived effects data from the site under investigation.

3.1.4 Suitability for In-Place Pollutant Control--

The guidance document was developed specifically for the assessment of in-place pollutant problems. It is designed to fit into the framework of evaluating and choosing remedial options by providing an adequate database upon which to base such decisions. A companion document (IJC 1988b) provides guidance in the selection of courses of remediation.

3.1.5 Suitability for Source Control--

The detailed assessment provides an adequate framework for identifying hot spots, and for establishing significant differences from background conditions. In some cases, the resultant maps may provide further evidence of contaminant sources and migration patterns, using spatial autocorrelation techniques. Presumably, such evidence could facilitate regulation of identified sources. However, source control is not a primary objective of the IJC (1988a) strategy.

3.1.6 Suitability for Disposal Applications--

Although the document was not intended for the use in decision-making related to the disposal of material from navigational dredging, the data generated from an initial assessment could be used to make initial disposal decisions. Other practices for the assessment of dredged material may be more cost-effective, however.

3.2 General Advantages and Limitations

3.2.1 Ease of Use--

The proposed strategy is designed to be applicable to the AOC under investigation. It is intended to flexible, relying on the judgment and experience of those who apply it. A detailed assessment would only be practical in cases where a major remedial effort is contemplated.

3.2.2 Relative Cost--

The Stage I and II assessments are costly compared to other less comprehensive methods of assessing sediment quality. However, when compared to the potential remedial costs, the assessment costs are relatively small. The sequential approach is designed to reduce sampling, analysis, and expense where possible. In many cases, the Stage I assessment need not be done. If it is clear that a sediment contamination problem exists, then the investigators may proceed directly to Stage II assessment. Alternatively, if the Stage I assessment produces no results of concern, then Stage II need not be undertaken. The cost of a detailed assessment, although relatively high, is controlled somewhat by the sequential approach to data collection. No firm cost figures are currently available, but assessments planned for priority AOCs under Section 118(c)(3) of the Water Quality Act of 1987 are projected to cost in the range of $500,000. These costs are expected to vary from site to site.

3.2.3 Tendency to be Conservative--

The strategy is designed to be highly protective of the environment. It combines chemical analysis, toxicity testing, and examination of indigenous communities to ensure that no significant effects are overlooked. Because the application of criteria is left to the expert judgment of the investigator, the degree of conservatism in decision-making will be variable.

3.2.4 Level of Acceptance--

The guidance document (IJC 1988a) does not describe a new method, but rather a combination of several types of methods, each widely accepted in its own sphere. The strategy as a whole is being used for the first time in 1989.

3.2.5 Ability to be Implemented by Laboratories with Typical Equipment and Handling Facilities--

None of the methods is particularly unusual or difficult, but the detailed assessment requires a breadth of expertise and resources that an individual organization may not possess. The strategy will need to be implemented by drawing upon a variety of expertise in a given geographical area.

3.2.6 Level of Effort Required to Generate Results--

The total level of effort for a detailed assessment will be relatively high in most cases. This strategy is most suitable for major evaluation projects.

3.2.7 Degree to which Results Lend Themselves to Interpretation--

The actual statistical analysis and interpretation to generate effects conclusions are relatively complex, and should be done only by trained investigators. Specific statistical protocols are not recommended. However, the reader is given an array of choices, with comments on their respective strengths and weaknesses. The ultimate decision is left to the investigator. The inclusion of chemical, toxicological, and infaunal information in the database allows the investigator to compare different types of indicators before making decisions.

3.2.8 Degree of Environmental Applicability--

One of the strengths of a strategy that includes *in situ* community analysis is that effects data have a high degree of environmental relevance.

Site-relevant species can even be substituted in the bioassay battery if necessary, and the body burden and community structure data are always site-specific.

3.2.9 Degree of Accuracy and Precision--

The strategy proposed by the IJC (1988a) is not a single method, but rather guidance for a study design containing many options and decision points. Overall precision or accuracy values would be impossible to calculate. Nevertheless, the criteria for selecting recommended protocols included a consideration of attainable precision. In many sections, the investigator is directed to choose the required level of precision for a given measurement during the study design process. The "accuracy" of an integrated strategy is difficult to assess, but the methods recommended by the IJC (1988a) were chosen for their relevance to the Great Lakes ecosystem.

4.0 STATUS

4.1 Extent of Use

IJC's (1988a) document was published in December 1988, and distributed in early 1989. The strategy is intended for the Great Lakes, and will be used for the first time in 1989. Most of the individual methods recommended are widely used and accepted.

4.2 Extent to Which the Approach Has Been Field-Validated

The first extensive field validation of the approach will take place in 1989-1991 as part of the ARCS program under Section 118(c)(3) of the Water Quality Act of 1987.

4.3. Reasons for Limited Use

Most component protocols are in wide use. Because the IJC (1988a) document has only recently appeared, it has not yet been applied.

4.4 Outlook for Future Use and Development

With the backing of both signatories to the Great Lakes Water Quality Agreement, the document seems destined for widespread use in the Great Lakes basin. As methods will progress, the document will be updated in each of its sections.

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